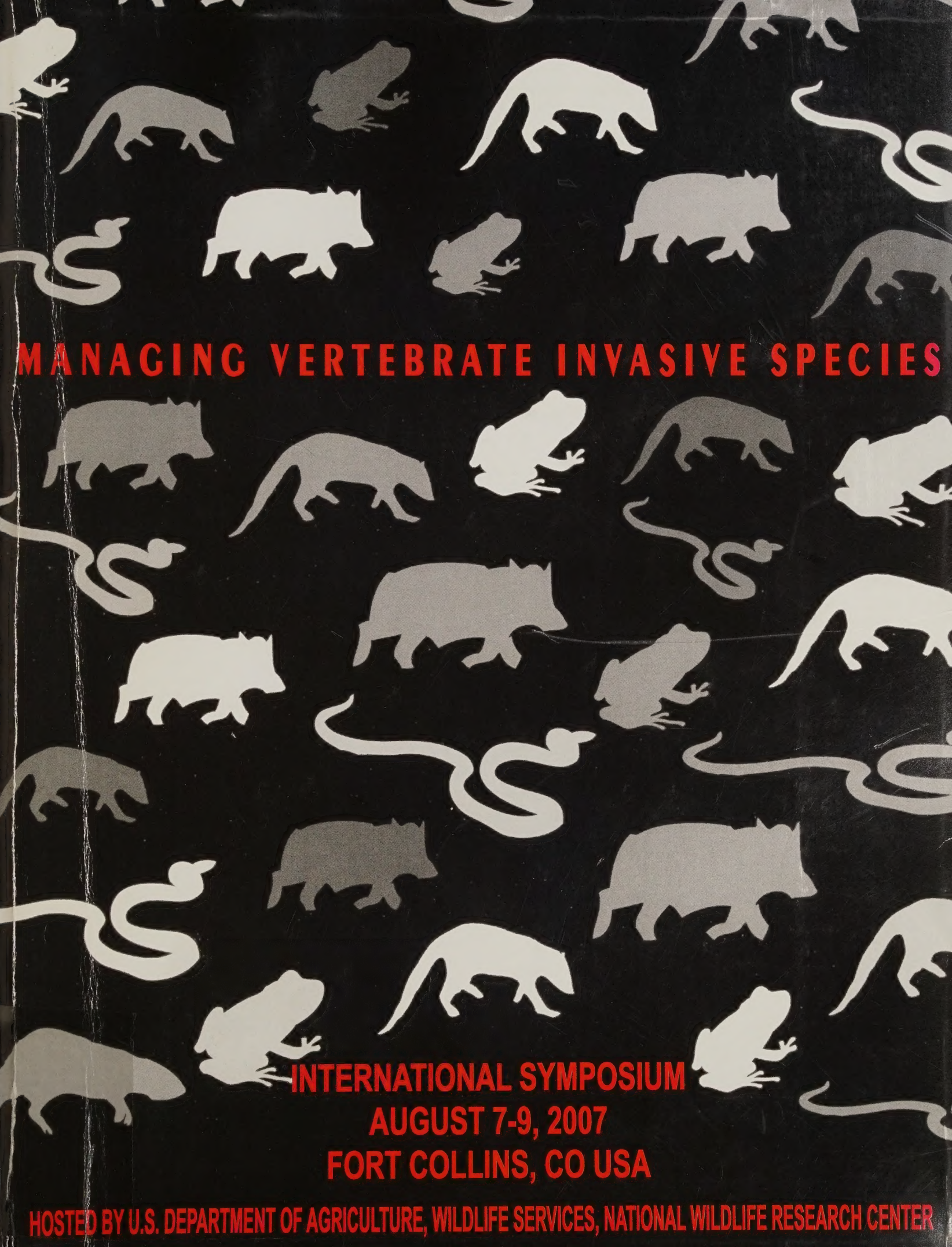
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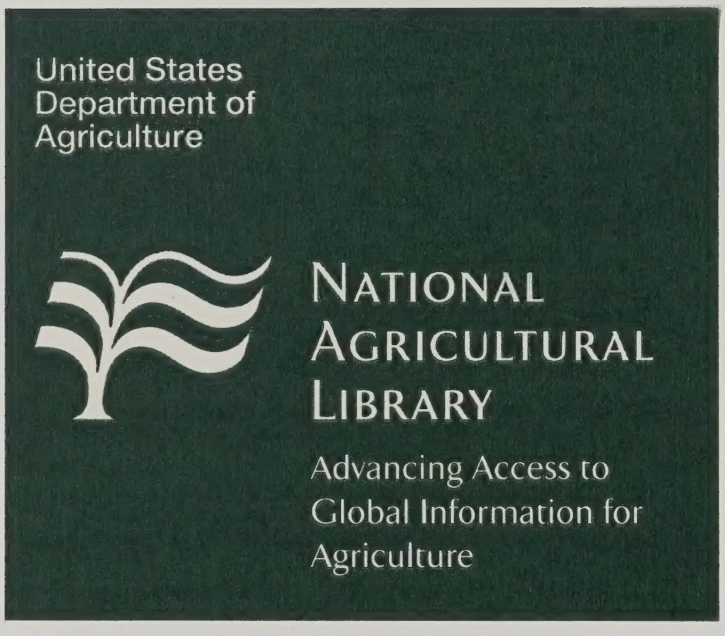
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# MANAGING VERTEBRATE INVASIVE SPECIES

## INTERNATIONAL SYMPOSIUM
## AUGUST 7-9, 2007
## FORT COLLINS, CO USA

HOSTED BY U.S. DEPARTMENT OF AGRICULTURE, WILDLIFE SERVICES, NATIONAL WILDLIFE RESEARCH CENTER




*Cover art by Anna Downey, Colorado State University.*

# MANAGING VERTEBRATE INVASIVE SPECIES:

# PROCEEDINGS OF AN INTERNATIONAL SYMPOSIUM

**Fort Collins, Colorado**

**August 7-9, 2007**

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CATALOGING PREP

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# FOREWORD

The effects of invasive vertebrate species on agriculture, human health and safety, and the environment are a growing concern around the world. The number of incidents of invasive species causing harm continues to climb with increased worldwide travel and transportation of goods. The focus for many decades was on invasive pathogens, plants, and invertebrates because they can greatly affect human and animal health and food supplies. In recent years, invasive vertebrate species, such as rats, feral pigs, and feral cats, have garnered more attention because the magnitude of their impacts have been repeatedly highlighted in the media. In response, better methods of prevention, detection, and management of invasive species have been developed. For example, several eradications of invasive vertebrates on islands have been successful, although management on mainland settings is generally much more challenging.

This volume is the outcome of an international symposium on managing vertebrate invasive species. The symposium was organized and sponsored by the National Wildlife Research Center of the U.S. Department of Agriculture's Animal and Plant Health Inspection Service, Wildlife Services. The symposium was held in Fort Collins, Colorado, on August 7-9, 2007, and was attended by researchers, managers, agency and military personnel, academicians, private industry employees, and representatives of conservation groups and non-governmental organizations. Participants were from the United States, Canada, Chile, United Kingdom, Australia, and New Zealand, as well as several island territories in the Pacific and Caribbean.

As readers will discover, presentations, both oral and poster, were wide-ranging in their coverage of species and problem situations. Plenary speakers set the stage with their talks on the scale of the issue, national and international perspectives, and global initiatives. Other speakers addressed public education, economics, and invasion prevention. These were followed by sessions organized by taxonomic groups: reptiles and amphibians, rodents, other mammals, and birds and multiple species. Speakers represented a wealth of knowledge and experience, and we saw an impressive exchange of ideas and substantial collaboration-building.

We hope this proceedings will serve not only as an overview of the current knowledge about vertebrate invasive species and method and strategy development, but also as a catalyst and guide for future research and management in this rapidly growing area of wildlife damage management. We look forward to the development and practical application of new prevention, detection, management and eradication methods and strategies, and foresee an increasing reliance on these emerging wildlife management tools derived from partnerships among academic, governmental, and private sectors.

Gary W. Witmer
William C. Pitt
Kathleen A. Fagerstone

Editors

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## WELCOME AND OPENING REMARKS

RICHARD L. BRUGGERS, Director, National Wildlife Research Center, USDA, APHIS, Wildlife Services, Fort Collins, Colorado, USA

Greetings and welcome to Fort Collins. We at the National Wildlife Research Center (NWRC) have had the privilege of organizing and hosting four international symposia since 1993. The first three were on the topics of contraception and wildlife, repellents in wildlife management, and the economics of wildlife damage management.

Today, it is my pleasure to welcome you to our fourth symposium: the International Symposium on Managing Vertebrate Invasive Species. I know our Symposium's organizing committee has strived to bring you a diverse group of international speakers....all of whom are working on the cutting edge of invasive species research, management, and eradication. More specifically, much work is being done in the areas of early detection, rapid response, biological and economic impacts, prevention and control, resource recovery, public education and support, and global initiatives.

Before I share with you a little background on the mission of the National Wildlife Research Center and our reasons for hosting this important symposium, I would like to recognize the members of the Symposium's organizing committee who helped make this event possible: Drs. Kathy Fagerstone, Gary Witmer, Will Pitt, Mark Tobin, and Tyler Campbell; Stephen Kendrot, Robert Myers, John Sinclair, Diana Dwyer, and Gail Keirn. These individuals are here to ensure the Symposium runs smoothly and to assist you in any way. Please feel free to contact any of them for assistance while at the Symposium or in the future. I would also like to acknowledge and thank the Colorado State University's Conference Services for assistance with registration and coordination with the Hilton Hotel staff. The primary sponsors for this Symposium are the U.S. Department of Agriculture's Animal and Plant Health Inspection Service and the National Invasive Species Council.

The USDA's National Wildlife Research Center is headquartered here in Fort Collins on the Colorado State University's Foothills Research Campus. The Center also has eight field stations located throughout the United States. Five of these field stations are actively involved in invasive species research (Hilo, Hawaii; Gainesville, Florida; Starkville, Mississippi; Bismarck, North Dakota; and Kingsville, Texas). The NWRC is the federal institution that provides national leadership for research and information to resolve human-wildlife conflicts. The Center applies scientific expertise to the development of methods and information while maintaining the quality of the environment shared with wildlife to resolve human-wildlife conflicts related to agriculture (crops, livestock, aquaculture, and timber), human health and safety (wildlife diseases, aircraft-bird strikes), property, threatened and endangered species protection, and invasive species management. The U.S. Department of Agriculture has been involved with invasive species management and research for quite some time. Recently, we completed construction of a 25,000-square-foot vertebrate invasive species research building in Fort Collins. This building will help us carry out the invasive species mandate of our APHIS mission. The building enhances our opportunities and abilities to study the biology, behavior and physiology of a variety of invasive wildlife species and to develop new methods of management and control. We are conducting research in our new invasive species building with brown treesnakes, Norway rats, roof rats, Polynesian rats, house mice, Gambian giant pouched rats, monk parakeets, Burmese pythons, and Nile monitor lizards. For additional information on the research conducted by the NWRC, please go to our website: www.aphis.usda.gov/wildlife_damage/nwrc/

In closing, I would like to note that this Symposium provides all of us with excellent opportunities to exchange ideas, build new partnerships, discuss and debate the effectiveness of existing tools and techniques, and to work together towards new initiatives involving invasive species. I wish you all a very thought-provoking, productive and fun week.

# ENVIRONMENTAL AND ECONOMIC COSTS OF VERTEBRATE SPECIES INVASIONS INTO THE UNITED STATES

DAVID PIMENTEL, College of Agriculture and Life Sciences, Cornell University, Ithaca, New York, USA

***Abstract:*** The more than 50,000 species of plants, animals, and microbes introduced into the United States (US) cause more extinction of native species than most any other threat and cause more than $120 billion in damages and control costs each year. In particular, invasive mammals and birds cause major environmental and economic losses totaling about $46 billion per year in damage and control costs per year. Feral cats, rats, and hogs are especially serious pests. Pigeons and starlings are a major concern in cities and US agriculture. Prevention and management strategies will be discussed.

***Key Words:*** alien species, control costs, economics, environment, invasive species, management strategies, pests, vertebrates.

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## INTRODUCTION

Of the estimated 750,000 species of organisms in the United States (US), an estimated 50,000 species of plants, animals, and microbes have been introduced into the nation (Pimentel et al. 2000). Most of the vertebrate and plant species that are invaders were intentionally introduced, whereas most of the microbes and invertebrates that have invaded the US were accidentally introduced.

The total damage and control costs from all invading species in the US are estimated to be about $120 billion per year (Pimentel 2005). Most of the benefits come from the 99% of crops and livestock that were intentionally introduced (USCB 2007). The benefit from the agricultural system alone is estimated to be about $800 billion per year (USCB 2007).

The focus of this article will be on vertebrate species that were introduced into the US (Table 1). As mentioned, most of these species were intentionally introduced.

## MAMMALS

About 28 species of mammals have been introduced into the US; these include dogs (*Canis familiaris*), cats (*Felis catus*), horses (*Equus caballus*), burros (*E. asinus*), cattle (*Bos Taurus*), sheep (*Ovis aries*), pigs (*Sus scrofa*), goats (*Capra hirus*), and deer (*Cervus spp.*) (Drost and Fellers 1995, Layne 1997). Several of these species have escaped or were released into the wild; some have become pests by preying on native animals, grazing on vegetation, or intensifying soil erosion. For example, goats introduced on San Clemente Island, California, are responsible for the extinction of 8 endemic plant species as well as the endangerment of 8 other native plant species (Kurdila 1995).

Many small mammals have also been introduced into the US. These species include a number of rodents, (black or ship rat [*Rattus rattus*], Norway or brown rat [*Rattus norvegicus*], house mouse [*Mus musculus*], and European rabbit [*Oryctolagus cuniculus*], Layne 1997).

Some introduced rodents have become serious pests on farms, in industries, and in homes (Layne 1997). Rats and mice are particularly abundant and destructive on farms. On poultry farms, there is approximately 1 rat per 5 chickens (D. Pimentel, unpublished data, Smith 1984). Using this ratio, the total rat population on US poultry farms may easily number more than 1.8 billion (USDA 2006). Assuming that the number of rats per chicken has declined because of improved rat control since these observations were made, I estimate that the number of rats on poultry and other farms is approximately 1.5 billion. With an estimated additional 1 rat per person in homes and related areas (Wachtel and McNeely 1985, Discover 2006), there are an estimated 300 million introduced rats in the US (USCB 2007).

If we assume, conservatively, that each adult rat consumes or destroys stored grains (Chopra 1992, Ahmed et al. 1995) and other materials valued at $15/yr, then the total cost of destruction by introduced rats in the US is more than $27 billion per year. In addition, rats cause fires by gnawing

**Table 1.** Vertebrate animal introductions into the US and the total estimated damage and control costs caused by these animals.

| Category | Species | Damage and Control Costs (x $1 million) |
|---|---|---|
| Mammals | Wild horses and burros | $5 |
| | Feral pigs | $1,500 |
| | Mongooses | $50 |
| | Rats | $27,000 |
| | Cats | $14,000 |
| | Dogs | $620 |
| Birds | Pigeons | $1,400 |
| | Starlings | $800 |
| Reptiles & amphibians | Brown treesnakes | $5.6 |
| Fish | | $1,000 |
| Total | | $46,380.6 |

electric wires, pollute foodstuffs, and act as vectors of several diseases, including salmonellosis and leptospirosis, and, to a lesser degree, plague and murine typhus (Richards 1989). They also prey on some native invertebrate and vertebrate species like birds and bird eggs (Amarasekare 1993, Woodworth and Fancy 2006).

One of the first cases of the failure of biological control is the use of the Indian mongoose (*Herpestes auropunctatus*). It was first introduced into Jamaica in 1872 for biological control of rats in sugarcane (Pimentel 1955). It was subsequently introduced to the territory of Puerto Rico, other West Indian Islands, and Hawaii for the same purpose. In some cases, the mongoose controlled the black rat, but not the Norway rat, and it preyed heavily on native ground nesting birds (Pimentel 1955, Vilella and Zwank 1993). It also preyed on beneficial native amphibians and reptiles, causing at least 7 amphibian and reptile extinctions in Puerto Rico and other islands of the West Indies (Henderson 1992). In addition, the mongoose emerged as the major vector and reservoir of rabies and leptospirosis in Puerto Rico and other islands (Everard and Everard 1992). Based on public health damages, killing of poultry in Puerto Rico and Hawaii, extinctions of amphibians and reptiles, and destruction of native birds, it is estimated that the mongoose is causing approximately $50 million in damages each year in Puerto Rico and the Hawaiian Islands (Pimentel et al. 2000).

Introduced cats have also become a serious threat to some native birds and other animals in the US There are an estimated 67 million pet cats in the US (USCB 2007), plus as many as 30 million feral cats (Luoma 1997). Cats prey on native birds (Fitzgerald 1990), plus small native mammals, amphibians, and reptiles (Dunn and Tessaglia 1994). Estimates are that feral cats in Wisconsin and Virginia kill more than 3 million birds in each state per year (Luoma 1997). Based on the Wisconsin and Virginia data, I assume that 5 birds are killed per feral cat/year; McKay (1996) reports that pet cats kill a similar number of birds as feral cats. Thus, about 470 million birds are killed by cats per year in the nation. Each adult bird can be valued at $30. This cost per bird is based on the literature that reports that a bird watcher spends $0.40 per bird observed, a hunter spends $216 per bird shot, and specialists spend $800 per bird reared for release; in addition, note that EPA fines polluters $10 per fish killed, including small, immature fish (Pimentel and Greiner 1997). Therefore, the total damage to US bird population is approximately $14 billion/yr. This figure does not include small mammals, amphibians, and reptiles that are killed by feral and pet cats (Dunn and Tessaglia 1994).

Like cats, most dogs introduced into the US were introduced for domestic purposes, but some have escaped into the wild. Many of these wild dogs run in packs and kill deer, rabbits, and domestic cattle, sheep, and goats. Carter (1990) reported that feral dog packs in Texas cause more than $5 million in livestock losses each year. Dog packs have also become a serious problem in Florida (Layne 1997). In addition to the damages caused by dogs in Texas, and conservatively assuming $5 million for all damages for the other 49 states combined, total losses in livestock kills by

dogs per year would be approximately $10 million per year.

Moreover, approximately 4.7 million people are bitten by feral and pet dogs annually, with 800,000 cases requiring medical treatment (Sacks et al. 1996). Centers for Disease Control estimates medical treatment for dog bites costs $165 million/yr, and the indirect costs, such as lost work, increase the total costs of dog bites to $250 million/yr (Colburn 1999, Quinlan and Sacks 1999). In addition, dog attacks cause between 11 and 14 deaths per year, and 80% of the victims are small children (CDC 1997). The total damage and control costs associated with dogs are estimated to be $620 million (Pimentel et al. 2005).

Horses and burros, deliberately released in the western US, have attained wild populations of approximately 50,000 animals (Pogacnik 1995). These animals graze heavily on native vegetation, allowing non-indigenous annuals to displace native perennials (Rosentreter 1994). Burros inhabiting the northwestern US also diminish the primary food sources of native bighorn sheep and seed-eating birds, thereby reducing the abundance of these native animals (Kurdila 1995). In general, the large populations of introduced wild horses and burros cost the nation an estimated $5 million/yr in forage losses (Pimentel et al. 2000).

Feral pigs, native to Eurasia and North Africa, have been introduced into some US parks for hunting, including parks in the California coastal prairie and Hawaiian Islands, where they have substantially changed the vegetation in these parks (Kotanen 1995). In Hawaii, more than 80% of the soil is bare in regions inhabited by pigs (Kurdila 1995). This disturbance allows annual plants to invade the overturned soil and intensifies soil erosion. Pig control per park in Hawaii (approximately 1,500 pigs/park) (Stone et al. 1992) costs about $150,000/yr. Assuming that the 3 parks in Hawaii have similar pig control problems, the total is $450,000/yr (R. Zuniga, Cornell University, personal communication).

Feral pigs have also become a serious problem in Florida and Texas where their populations have risen to about 2 million in each state (Giuliano and Tanner 2007, Mapston 2007). Another 1 million hogs are estimated to be in Hawaii, California, and other southern US. On the Edwards Plateau and Rolling Plains region the damages to agriculture average greater than $10,000 per landowner (Texas Association of Counties 2003). In other areas of Texas, the damage ranges between $1,800 and $6,000. In addition, landowners spend an average of $2,631 for control and/or repairs (Texas Association of Counties 2003).

In Texas, and elsewhere, pigs damage grain, peanut, soybean, cotton, hay, and various vegetable crops, and the environment (Rollins 1998). Pigs also transmit and are reservoirs for 30 viral and bacterial diseases of livestock and human diseases, including brucellosis, pseudobrucellosis, and trichinosis (Davis 1998, Williams and Barker 2001).

Nationwide, as mentioned, there are an estimated 5 million feral pigs. Based on crop damages and control costs of about $300 per pig annually, feral pigs in the US are causing at least $1.5 billion in damages and control costs.

## BIRDS

Approximately 100 of the 1,000 bird species in the US are exotic (Temple 1992, EPA 2005). Of the approximately 100 introduced bird species, only 5%, including chickens, are considered beneficial. Most (56%), though, are considered pests (Temple 1992). Pest species include the common pigeon (*Columba livia*), which was introduced into the US for agricultural purposes.

Introduced bird species are an especially severe problem in Hawaii. A total of 54 of the 142 non-indigenous bird species introduced since 1850 in Hawaii are still extant on the islands (Moulton and Pimm 1983, Pimm 1991, Earlham College 2002). One such species, the common myna (*Acridotheres tristis*), was introduced to help control pest cutworms and armyworms in sugarcane (Kurdila 1995). However, it became the major disperser of seeds of an introduced pest weed, *Lantana camara*. In the continental US, the English house sparrow (*Passer domesticus*) was introduced in 1853 to control the canker worm (Laycock 1966, Roots 1976). By 1900, they had become pests because they damage plants around homes and public buildings and consume wheat, corn, and the buds of fruit trees (Laycock 1966). Furthermore, English sparrows harass native birds, including robins (*Turdus migratorius*), Baltimore orioles (*Icterus galbula*), yellow-billed cuckoos (*Coccyzus americanus*), and black-billed cuckoos (*C. erythropthalmus*), and displace native bluebirds (*Sialia* spp.), nuthatches (*Sitta* spp.), black-capped chickadees (*Parus* spp.), house wrens (*Troglodytes aldon*), tree swallows (*Iridoprocne bicolor*), tufted titmice (*Parus bicolor*), purple martins (*Progne subis*), and cliff swallows (*Petrochelidon pyrrhonota*) from their nesting sites (Laycock 1966,

Roots 1976, Long 1981, Purple Martin Society 2007). They are also associated with the spread of about 29 human and livestock diseases (Weber 1979).

The single-most serious pest bird in the US is the exotic common pigeon that exists in most cities of the world, including those in the US (Robbins 1995). Pigeons are considered a nuisance because they foul buildings, statues, cars, and sometimes people, and feed on grain (Long 1981, Smith 1992). The control costs of pigeons are at least $9 per pigeon per year (Haag-Wackernagel 1995). Assuming 1 pigeon per ha in urban areas (Johnston and Janiga 1995) or approximately 0.5 pigeons per person, and using potential control costs as a surrogate for losses, pigeons cause an estimated $2 billion/yr in damages. These control costs do not include the environmental damages associated with pigeons, which serve as reservoirs and vectors for over 50 human and livestock diseases, including parrot fever, ornithosis, histoplasmosis, and encephalitis (Weber 1979, Long 1981).

Some birds threaten crop production. European starlings (*Sturnus vulgaris*) are serious pests and are estimated to occur at densities of more than 1 per ha in agricultural regions (Moore 1980). Starlings are capable of destroying as much as $2,000 worth of cherries per hectare (Feare 1980). In grain fields, starlings consume about $6/ha of grain (Feare 1980). Conservatively assuming $5/ha for all damages to several crops in the US, the total loss due to starlings would be approximately $800 million/yr. In addition, these aggressive birds have displaced numerous native birds (Laycock 1966). Starlings have also been implicated in the transmission of 25 diseases, including parrot fever and other diseases of humans (Laycock 1966, Weber 1979).

## AMPHIBIANS AND REPTILES

Amphibians and reptiles introduced into the US number about 53 species. All these non-indigenous species occur in relatively warm states. Florida is now host to 30 species and Hawaii to 12 (McCoid and Kleberg 1995, Lafferty and Page 1997). The negative ecological impacts of several of these exotic species have been enormous.

The brown treesnake (*Boiga irregularis*) was accidentally introduced to the snake-free US territory of Guam immediately after World War II, when military equipment was moved onto Guam (Fritts and Rodda 1995). Soon the snake population reached densities of 100 per ha, and dramatically reduced native bird, mammal, and lizard populations. Of the 13 species of native forest birds originally found on Guam, only 3 still exist (Rodda et al. 1997); of the 12 native species of lizards, only 3 have survived (Rodda et al.1997). The snake eats chickens, eggs, and caged birds, causing major problems to small farmers and pet owners. It also crawls up trees and utility poles and has caused power outages on the island. One island-wide power outage caused by the snake cost the power utility more than $250,000 (Teodosio 1987). Local outages that affect businesses are estimated to cost from $2,000 to $10,000 per commercial customer per year (Coulehan 1987). With about 86 outages per year (BTSCP 1996), my estimate of the cost of snake-related power outages is conservatively $1 million/yr.

In addition, the brown treesnake is slightly venomous, and has caused public health problems, especially when it has bitten children. At one hospital emergency room, about 26 people per year are treated for snake bites (OTA 1993). Some bitten infants require hospitalization and intensive care, at an estimated total cost of $25,000 per year.

The total costs of endangered species recovery efforts, environmental planning related to snake containment on Guam, and other programs directly stemming from the snake's invasion of Guam reach more than $1 million per year; in addition, up to $2 million per year is invested in research to control this serious pest. The brown treesnake has also invaded Hawaii but thus far has been exterminated. Hawaii's concern about the snake, though, has prompted the federal government to invest $1.6 million per year in brown treesnake control (Holt 1997-1998). The total cost associated with the snake is therefore more than $5.6 million/yr.

## FISH

A total of at least 152 non-indigenous fish species has been introduced into the US (Courtenay 1993, 1997, Fuller et al. 2005). Most of these introduced fish have been established in states with mild climates, such as Florida (50 species) (Courtenay 1997) and California (56 species) (Dill and Cordone 1997). In Hawaii, 33 non-indigenous freshwater fish species have become established (Maciolek 1984). Forty-four native species of fish are threatened or endangered in the US by non-indigenous fish species (Wilcove and Bean 1994). An additional 27 native fish species are also negatively affected by introductions (Wilcove and Bean 1994, Fuller et al. 2005).

Introduced fish species frequently alter the ecology of aquatic ecosystems. For instance, the grass carp (*Ctenopharyngodon idella*) reduces natural aquatic vegetation, while the common carp (*Cyprinus carpio*) reduces water quality by increasing turbidity. These changes have caused the extinctions of some native fish species (Taylor et al. 1984).

Although some native fish species are reduced in numbers, are driven to extinction, or hybridized by non-indigenous fish species, alien fish do provide some economic benefits in the improvement of sport fishing. Sport fishing contributes $69 billion to the economy of the US (Bjergo et al. 1995, Pimentel 2005, USCB 2007). However, even taking into account these economic benefits, based on the more than 40 non-indigenous species that have negatively affected native fishes and other aquatic biota, a conservative estimate puts the economic losses due to exotic fish at more than $1 billion annually. The benefit of the Great Lakes Fishery alone is estimated to be $4.5 billion per year (Pimentel 2005).

## CONCLUSION

With more than 50,000 introduced species of plants, animals, and microbes in the US, only a portion of these cause significant damage to agriculture, forestry, and natural ecosystems, and require costly control measures. Vertebrates are one group of invasive species that cause significant ecological and environmental damages. My best estimate for the annual cost of vertebrate species in the US is approximately $46 billion.

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## LITERATURE CITED

AHMED, E., I. HUSSAIN, AND J. E. BROOKS. 1995. Losses of stored foods due to rats at grain markets in Pakistan. International Biodeterioation & Biodegradation 36:125-133.

AMARASEKARE, P. 1993. Potential impact of mammalian nest predators on endemic forest birds of western Mauna Kea, Hawaii. Conservation Biology 7:316-324.

BJERGO, C., C. BOYDSTUN, M. CROSBY, S. KOKKANAKIS, AND R. SAYERS. 1995. Non-native aquatic species in the United States and coastal water. Pages 428-430 *in* E. T. LaRoe, G. S. Farris, C. E. Puckett, P. D. Doran, and M. J. Mac, editors. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, D.C., USA.

BTSCP. 1996. Brown tree snake control plan. Brown Tree Snake Control Committee, Aquatic Nuisance Species Task Force, June 1996, Honolulu, Hawaii, USA.

CARTER, C. N. 1990. Pet population control: another decade without solutions? Journal of American Veterinary Medicine Association 197:192-195.

CDC. 1997. Dog-bite-related fatalities – United States, 1995-1996. Mortality and Morbidity Weekly Report of the Communicable Disease Center 46:463-467.

CHOPRA, G. 1992. Poultry farms. Pages 309-330 *in*. I. Prakash and P. K. Ghosh, editors. Rodents in Indian agriculture. Scientific Publishers, Jodhpur, India.

COLBURN, D. 1999. Dogs take a big bite out of health care costs. The Washington Post, February 2, 1999. Page z5.

COULEHAN, K. 1987. Powerless again. About your partners in business: snakes and GPA. Guam Business News. January 1987:13-15.

COURTENAY, W. R. 1993. Biological pollution through fish introductions. Pages 35-62 *in* B. N. McKnight, editor. Biological pollution: the control and impact of invasive exotic species, Indiana Academy of Science, Indianapolis, Indiana, USA.

COURTENAY, W. R. 1997. Non-indigenous fishes. Pages 109-122 *in* D. Simberloff, D. C. Schmitz, and T. C. Brown, editors. Strangers in Paradise, Island Press, Washington, D.C., USA.

COURTENAY, W.R., D. P. JENNINGS, AND J. D. WILLIAMS. 1991. Appendix 2. Exotic fishes of the United States and Canada. *In*: C.R. Robins, editor. A list of common and scientific names of fishes from the United States and Canda. Special Publication 20. American Fisheries Society, Bethesda, Maryland, USA.

DAVIS, D. S. 1998. Feral hogs and disease: implications for humans and livestock. Department of Veterinary Pathology, Texas A & M University, College Station, Texas, USA.

DILL, W. A, AND A. J. CORDONE. 1997. History and status of introduced fishes in California, 1871-1996. Fish Bulletin 178. The Resources Agency, Department of Fish and Game, State of California.

DISCOVER. 2006. Discover, science, technology, and the future. http://discovermagazine.com/2006/dec/20-things-rats (5/19/07).

DROST, C. A., AND G. M. FELLERS. 1995. Non-native animals on public lands. Pages 440-442 *in* E. T. LaRoe, G. S. Farris, C. E. Puckett, P. D. Doran, and M. J. Mac, editors. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals and ecosystems. U.S.

Department of the Interior, National Biological Service, Washington, D.C., USA.

DUNN, E. H., AND D. L. TESSAGLIA. 1994. Predation of birds at feeders in winter. Journal of Field Ornithology 65:8-16.

EARLHAM COLLEGE. 2002. Introduced species in Hawaii. Senior Seminar. Earlham College, Richmond, Indiana, USA.

EPA. 2005. Final list of bird species to which the Migratory Bird Treaty Act does not apply. Federal Register March 15, 2005 70:49.

EVERARD, C. O. R., AND J. D. EVERARD. 1992. Mongoose rabies in the Caribbean. Annals of the New York Academy of Sciences 653:356-366.

FEARE, C. J. 1980. The economics of starling damage. Pages 39-55 *in* E. N. Wright, I. R. Inglis, and C. J. Feare, editors. Bird problems in agriculture. The British Crop Protection Council, Croydon, United Kingdom.

FITZGERALD, B. M. 1990. Diet of domestic cats and their impact on prey populations. Pages 123-150 *in* D. C. Turner, and P. Bateson, editors. The domestic cat: the biology of its behavior, Cambridge University Press, Cambridge, United Kingdom.

FRITTS, T. H., AND G. H. RODDA. 1995. Invasions of the brown tree snake. Pages 454-456 *in* E. T. LaRoe, G. S. Farris, C. E. Puckett, P. D. Doran, and M. J. Mac, editors. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, D.C., USA.

FULLER, P. L., L. G. NICO, AND J. D. WILLIAMS. 2005. Non-indigenous fishes introduced into inland waters of the United States. USGS. Gainesville, Florida. USA.

GIULIANO, W. M., AND G. W. TANNER. 2007. Ecology of wild hogs in Florida. IFAS Extension, University of Florida.

HAAG-WACKERNAGEL, D. 1995. Regulation of the street pigeon in Basel. Wildlife Society Bulletin 23:256-260.

HENDERSON, R. W. 1992. Consequences of predator introductions and habitat destruction on amphibians and reptiles in the post-Columbus West Indies. Caribbean Journal of Science 28:1-10.

HOLT, A. 1997-1998. Hawaii's reptilian nightmare. World Conservation 4/97 - 1/98:31-32.

JOHNSTON, R. F., AND M. JANIGA. 1995. Feral pigeons. Oxford University Press, New York, USA.

KOTANEN, P. M. 1995. Responses of vegetation to a changing regime of disturbance: effects of feral pigs in a California coastal prairie. Ecography 18:190-197.

KURDILA, J. 1995. The introduction of exotic species into the United States: there goes the neighborhood. Environmental Affairs 16:95-118.

LAFFERTY, K. D., AND C. J. PAGE. 1997. Predation of the endangered tidewater goby, *Eucyclogobius newberryi*, by the introduced African clawed frog, *Xenopus laevis*, with notes on the frog's parasites. Copeia 3:589-592.

LAYCOCK, G. 1966. The alien animals. Natural History Press, New York, USA.

LAYNE, J. N. 1997. Non-indigenous mammals. Pages 157-186 *in* D. Simberloff, D. C. Schmitz, and T. C. Brown, editors. Strangers in paradise, Island Press, Washington, D.C., USA.

LONG, J. L. 1981. Introduced birds of the world: the worldwide history, distribution, and influence of birds introduced to new environments. Universe Books, New York, USA.

LUOMA, J. R. 1997. Catfight. Audubon 99:85-90.

MACIOLEK, J. A. 1984. Exotic fishes in Hawaii and other islands of Oceania. Pages 131-161 *in* W. R. Courtenay and J. R. Stauffer, editors. Distribution, biology, and management of exotic fishes, Johns Hopkins University Press, Baltimore, Maryland, USA.

MAPSTON, M. E. 2007. Feral hogs in Texas. Texas Cooperative Extension. Wildlife Services.

MCCOID, M. J., AND C. KLEBERG. 1995. Non-native reptiles and amphibians. Pages 433-437 *in* E. T. LaRoe, G. S. Farris, C. E. Puckett, P. D. Doran, and M. J. Mac, editors. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, D.C., USA.

MCKAY, G. M. 1996. Feral cats in Australia: origin and impacts. Unwanted aliens? Australia's introduced animals. Nature Conservation Council of NSW. The Rocks, NSW, Australia. Museum Circular No. 57. The University of the State of New York, State Education Department.

MOORE, N. W. 1980. How many wild birds should farmland support? Pages 2-6 *in* E. N. Wright, I. R. Inglis, and C. J. Feare, editors. Bird problems in agriculture, The British Crop Protection Council, Croydon, United Kingdom.

MOULTON, M. P., AND S. L. PIMM. 1983. The introduced Hawaiian avifauna: biogeographic evidence for competition. The American Naturalist 121:669-690.

OTA. 1993. Harmful non-indigenous species in the United States. Office of Technology Assessment, United States Congress, Washington, D.C., USA.

PIMENTEL, D. 1955. The control of the mongoose in Puerto Rico. American Journal of Tropical Medicine and Hygiene 41:147-151.

PIMENTEL, D. 2005. Aquatic nuisance species in the New York State Canal and Hudson River systems and the Great Lakes Basin: an economic and environmental assessment. Environmental Management 35:692-701.

PIMENTEL, D., AND A. GREINER. 1997. Environmental and soci-economic costs of pesticide use. Pages 51-78 *in* D. Pimentel, editor. Techniques for reducing

pesticide use: economic and environmental benefits, John Wiley & Sons, Chichester, United Kingdom.

PIMENTEL, D., L. LACH, R. ZUNIGA, AND D. MORRISON. 2000. Environmental and economic costs of non-indigenous species in the United States. BioScience 50:53-65.

PIMENTEL, D., R. ZUNIGA, AND D. MORRISON. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52:273-288.

PIMM, S. L. 1991. The balance of nature? University of Chicago Press, Chicago, Illinois, USA.

POGACNIK, T. 1995. Wild horses and burros on public lands. Pages 456-458 *in* E. T. LaRoe, G. S. Farris, C. E. Puckett, P. D. Doran, and M. J. Mac, editors. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, D.C., USA.

PURPLE MARTIN SOCIETY. 2007. The Purple Martin Society. The simple logistics of attracting Purple Martins to your backyard. http://www.purplemartins.com/opm-online-mannual/pmol.c.htm (5/21/2007).

QUINLAN, K. P., AND J. J. SACKS. 1999. Hospitalizations for dog bite injuries. Centers for Disease Control. http://www.cdc.gov/ncipc/duip/hospital.htm (2/23/1999).

RICHARDS, C. G. J. 1989. The pest status of rodents in the United Kingdom. Pages 21-33 *in* R. J. Putman, editor. Mammals as pests. Chapman and Hall, London.

ROBBINS, C. S. 1995. Non-native birds. Pages 437-440 *in* E. T. LaRoe, G. S. Farris, C. E. Puckett, P. D. Doran, and M. J. Mac, editors. Our living resources: a report to the nation on the distribution, abundance, and health of U.S. plants, animals and ecosystems. U.S. Department of the Interior, National Biological Service, Washington, D.C., USA.

RODDA, G. H., T. H. FRITTS, AND D. CHISZAR. 1997. The disappearance of Guam's wildlife. BioScience 47:565-574.

ROLLINS, D. 1998. Statewide attitude survey on feral hogs in Texas. Texas Agricultural Extension Service.

ROOTS, C. 1976. Animal invaders. Universe Books, New York, USA.

ROSENTRETER, R. 1994. Displacement of rare plants by exotic grasses. Pages 170-175 *in* S. B. Monsen and S. G. Kitchen, editors. Proceedings of ecology and management of annual rangelands. U.S. Department of Agriculture Forest Service, Rocky Mountain Research Station, Washington, D.C., USA.

SACKS, J. J., M. KRESNOW, AND B. HOUSTON. 1996. Dog bites: how serious a problem? Injury Prevention 2:52-54.

SMITH, R. H. 1984. Producers need not pay startling "rodent tax" losses. Feedstuffs 56:13-14.

SMITH, R. H. 1992. Rodents and birds as invaders of stored-grain ecosystems. Pages 289-323 *in* D. S. Jayas, N. D. G. White, and W. E. Muir, editors. Books in soils, plants, and the environment: stored-grain ecosystems. Marcel Dekker, Inc., New York, USA.

STONE, C. P., L. W. CUDDIHY, AND T. TUNISON. 1992. Response of Hawaiian ecosystems to removal of pigs and goats. Pages 666-702 *in* Alien plant invasions on native ecosystems in Hawaii: management and research. University of Hawaii Cooperative National Park Studies Unit, Honolulu, Hawaii, USA.

TAYLOR, J. N., W. R. COURTENAY, AND J. A. MCCANN. 1984. Known impacts of exotic fishes in the continental United States. Pages 322-373 *in* W. R. Courtenay, and J. R. Stauffer, editors. Distribution, biology, and management of exotic fishes. Johns Hopkins University Press, Baltimore, Maryland, USA.

TEMPLE, S. A. 1992. Exotic birds, a growing problem with no easy solution. The Auk 109:395-397.

TEODOSIO, R. 1987. Tree snake brings Guam blackouts. Pacific Magazine 12:42.

TEXAS ASSOCIATION OF COUNTIES. 2003. Feral hogs and brush control are top studies. Texas Association of Counties, Austin, Texas, USA.

USCB. 2007. Statistical abstracts of the United States. Census Bureau of the United States, Washington, D.C., USA.

USDA. 2006. Agricultural statistics. U.S. Department of Agriculture, Washington, D.C., USA.

VILELLA, F. J., AND P. J. ZWANK. 1993. Ecology of the small Indian mongoose in a coastal dry forest of Puerto Rico where sympatric with the Puerto Rican nightjar. Caribbean Journal of Science 29:24-29.

WACHTEL, S. P., AND J. A. MCNEELY. 1985. Oh rats. International Wildlife 15:20-24

WEBER, W. J. 1979. Health hazards from pigeons, starlings and English sparrows: diseases and parasites associated with pigeons, starlings, and English sparrows which affect domestic animals. Thomson Publications, Fresno, California, USA.

WILCOVE, D. S., AND M. J. BEAN. 1994. The big kill: declining biodiversity in America's lakes and rivers. Environmental Defense Fund, Washington, D.C., USA.

WILLIAMS, E. S., AND I. K. BARKER. 2001. Infectious diseases of wild mammals. Iowa State University, Ames, Iowa, USA.

WOODWORTH, B. L., AND S. G. FANCY. 2006. Hakalau forest birds project. U.S. Geological Survey, Pacific Island Ecosystems Research Center. http://biology.usgs.gov/pierc/PLWoodworth2HAKALAU2.htm (6/29/07).

# INVASIVE SPECIES: A NATIONAL PERSPECTIVE AND THE NEED FOR A COORDINATED RESPONSE

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***Abstract:*** The multiple economic, environmental, and animal and human health impacts of invasive species pose complex challenges in policy formation and governmental coordination. The National Invasive Species Council (NISC) was established by an Executive Order in 1999 to provide coordination, planning and facilitate cooperation among the diverse federal agencies and to take a more comprehensive approach to invasive species. NISC, assisted by the nonfederal Invasive Species Advisory Committee (ISAC) has developed several tools and enhanced capacities to address invasive species. Additionally, NISC has helped place greater emphasis on essential prevention, early detection and rapid response, research, public outreach, and international cooperation efforts. Invasive vertebrates are an increasing concern for NISC and for state, tribal, and local governments and the private sector. The invasive vertebrates further complicate an already complex issue. Invasive vertebrates require an even greater level of coordination and understanding. Important questions such as, best practices for the humane control and removal and effective public outreach messages must be answered. NISC continues to emphasize the critical need for effective multi-sector coordination when addressing invasive vertebrates.

***Key Words:*** federal policy, invasive species, management, National Invasive Species Council, National Invasive Species Management Plan, United States.

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## INTRODUCTION

Invasive species affect all the United States (US) and other nations. Society pays a price for invasive species, in dollars, and in increased wildfire frequencies, degraded wildlife habitat, additional stress on endangered species, more extensive water shortages, and wildlife and livestock disease epidemics, and other environmental damage. In some cases, invasive species, such as West Nile Virus, harm human health directly. As global trade, travel, and transport expand to more areas, increase in volume, and become more rapid, the risk of additional invasive species also increases.

People have always moved organisms both intentionally and unintentionally. Many plants animals and microorganism are intentionally imported into the US. Most crops, livestock and pets are non-native species. Their benefits are obvious. Well managed livestock populations are examples of non-native species that are not invasive. Most non-native species are benign or beneficial. In some cases, invasive species are smuggled into the US and others arrive as hitchhikers on goods and conveyances. Even a single invasive species can cause great harm. An "invasive species" is defined in Executive Order 13112 as a species that is 1) non-native (or alien) to the ecosystem under consideration, and 2) whose introduction causes or is likely to cause economic or environmental harm or harm to human health.

It is difficult to estimate the total cost of invasive species. However, one estimate places it in the range of $100 billion annually (Pimentel et al. 2000). Data for individual species indicate significant costs. Formosan termite (*Coptotermes formosanus shiraki*) costs an estimated $300 million in property damage annually in New Orleans, Louisiana alone (E. Bordes, personal communication). Substantial efforts are focused on vertebrate invasive species such as: brown treesnakes (*Boiga irregularis*) in Guam, nutria (*Myocastor coypus*) and feral pigs (*Sus scrofa*) in numerous states. Burmese pythons (*Python molurus*) and Gambian giant pouched rats (*Cricetomys gambianus*) in Florida represent more recent vertebrate invasions in the US. Invasive species are a primary impacting factor for about half of the plants and animals currently federally-listed as endangered with extinction (Wilcove et al. 1998).

Executive Order 13112 on Invasive Species established the National Invasive Species Council (NISC). NISC is co-chaired by the Secretaries of Agriculture, Commerce and the Interior; and includes a total of 13 federal agencies and departments that have a role in invasive species. It also established the Non-Federal Invasive Species Advisory Committee (ISAC) to advise NISC. NISC provides national leadership on invasive species and works to increase the coordination and effectiveness of federal efforts. NISC also contributes to the work of local, state, and tribal governments and that of the private sector and makes recommendations for international cooperation. General NISC staff responsibilities include: staying current with the scientific literature; monitoring media coverage; tracking federal and state invasive species legislation and programs; maintaining a comprehensive invasive species meeting calendar; analyzing current and planned efforts; reporting NISC Plan progress; answering media and public inquires; representing NISC at meetings; hosting interdepartmental meetings; facilitating ISAC; working with the NISC website; publishing the NISC e-mail newsletter; and supporting decision makers.

## UPDATING THE NATIONAL PLAN

The 2001 National Invasive Species Management Plan (henceforth Plan, NISC 2001) was the first national plan to focuses on invasive species. Many, but not all, of the Plan action items have been accomplished. As called for in Executive Order 13112, the Plan is being revised. The revised 2008 Plan was released for public comment in December of 2007. This document will direct federal efforts to prevent and control invasive species over the next 5 years.

The 2008 Plan is focused on 5 strategic goals: prevention, early detection and rapid response, control and management, restoration, and organizational collaboration. To accomplish these goals, critical support for efforts such as research, data and information management, education and outreach, and international cooperation elements are included in pertinent sections of the 2008 Plan.

Each of the 5 strategic goals specifies on-going objectives and the long-term vision for success in that area. Under each goal, objectives describe what is to be accomplished over the next 5 years, and implementation tasks describe what agencies expect to do in order to accomplish that objective. Within the implementation tasks, performance elements describe how progress towards accomplishing identified tasks will be evaluated. Of course, the accomplishment of specific implementation tasks and performance elements depend upon agency budgets, and in some cases, legal or regulatory changes.

The current draft of the 2008 Plan requires the work of 35 different entities (typically agencies or bureaus with NISC members' departments and agencies) to achieve a total of 79 performance elements. Participating agencies are identified either as a "Lead" or a "Participant" to describe their role in accomplishing a specific performance element. Most (45) performance elements call for a finite accomplishment that once completed will require little continuing effort to coordinate. The remaining 34 performance elements require establishment and on-going coordination.

## LEADERSHIP AND COORDINATION

NISC works to provide national leadership and coordinate the efforts of the more than 30 federal agencies within the thirteen NISC members' departments that currently deal with invasive species. When appropriate, NISC also draws on existing organizations for coordination and leadership assistance, such as state invasive species councils (there are about 22 currently), regional organizations, Exotic Plant Pest Councils, Aquatic Nuisance Species Task Force (ANSTF), Federal Interagency Committee on the Management of Noxious and Exotic Weeds (FICMNEW), Federal Interagency Committee on Terrestrial Animals and Pathogens (ITAP), and other groups. States play a key role in the management of invasive species. The 2008 Plan emphasizes assisting states.

## PREVENTION

Prevention is the first-line of defense. It can be the most cost-effective approach to protectiuon against invasive species. Once a species becomes widespread, controlling the species may require significant and sustained expenditures. Therefore, public investment in prevention tools, resources and infrastructure are indispensable in protecting human health, agriculture and natural resources. Since the first 2001 Plan, progress on risk analyses, screening processes, evaluation of pathways, and the scientific understanding of the biology and ecology of invasive species has been made. A joint ANSTF/NISC Prevention Committee is currently addressing prevention issues.

## EARLY DETECTION AND RAPID RESPONSE

Even the best prevention efforts cannot stop all invasive species. Early detection, rapid assessment and rapid response (EDRR) increases the likelihood that localized invasive populations will be found, contained, and eradicated from areas before they become established. EDRR can slow range expansion, and reduce the need for extensive and costly long-term control efforts. Effective EDRR depends upon the timely ability to answer critical questions including: What is the species of concern, and has it been authoritatively identified? Where is it located and likely to spread? How important is it to respond (i.e., what is the potential harm the species may cause?) How can it be acted upon? Who has the authority and capacity to act upon it? How will the effort be funded? EDRR capacity at the regional, state and federal levels has improved, however, significant gaps remain.

## CONTROL AND MANAGEMENT

Eradication of well established, widespread invasive species, however, may not be feasible. Widespread invasive species are subject to long-term management efforts that slow the rate of range expansion and lessen the impacts of invasive populations. Invasive species can span geographic and jurisdictional boundaries. Their management requires regionally coordinated action. Information on the distribution, abundance, rates of spread, and impacts of invasive species is critical. Invasive plants, animals, and pathogens cause different impacts, as do aquatic invaders. Understanding their ecological, economic, and social impacts is important in prioritizing management operations. Differing management tools are needed to assess, remove and contain invasive species populations and guide management decisions. These tools are applied within coordinated and integrated invasive species management strategies that are adjusted, as needed. Financial and human resources limit the capacity to manage invasive species.

## RESTORATION

Executive Order 13112 requires federal agencies to "provide for restoration of native species and habitat conditions in ecosystems that have been invaded." Often plant and animal communities can recover following the control of the invasive species. However, the damage may be so extensive in some areas that further actions are required. Without restoration, some areas may become re-infested by the same or additional invasive species. NISC plans to further develop and issue recommendations, guidelines and monitoring procedures for federal land and water management agencies to use, where feasible, in restoration activities.

## INTERNATIONAL COOPERATION

The US cannot succeed in addressing its domestic invasive species problems unless it also works within the international arena and strategies that augment the capacity of other nations to manage invasive species. Once invasive species become established, they can threaten an entire region, trading partners, and countries along trade routes. NISC is working to strengthen and expand US participation in mutually supportive standards and international codes of conduct. It will also develop a strategy and support materials for US representatives that encourage and assist other countries with the development of coordinated policies and programs.

## CONCLUSION

Invasive species may at times seem overwhelming. However, considerable success is being achieved in the prevention, detection, eradication, and control of some invasive species. Additional research and information exchange, new detection and eradication techniques, and innovative control methodologies and collaborative models are increasing our capacity to solve invasive species problems. The 2008 National Invasive Species Management Plan takes a strategic approach and builds on existing programs to maximize federal efforts over the next 5 years to prevent and control invasive species in order to protect our environment, economy, and human health.

## LITERATURE CITED


National Invasive Species Council (NISC). 2001. Meeting the invasive species challenge: national invasive species management plan. NISC, Washington, D.C., USA.

Pimentel, D., L. Lach, R. Zuniga, and D. Morrison. 2000. Environmental and economic costs of nonindigenous species in the United States. BioScience 50:53-65.

Wilcove, D., D. Rothstein, J. DuBow, A. Phillips, and E. Losos. 1998. Quantifying threats to imperiled species in the United States. BioScience 48:607-615.

# INTERNATIONAL PERSPECTIVE ON PROGRAMMES OF INVASIVE VERTEBRATE SPECIES

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***Abstract:*** An account of global initiatives in invasive alien species, including vertebrates, was presented based on the author's experience and that of colleagues, mainly in the developing world. The account focused on those programmes which highlight typical problems or best practice for design and which promised improvements in the future. Some of the characteristics of these programmes were described and discussed. Techniques used in vertebrate control programmes, especially in developing island states, were also described with a commentary on their efficacy and suitability for use. Finally, some future directions were recommended in terms of the design of in-country and regional programmes and the methodology best suited for these programmes.

***Key Words:*** international programmes, international projects, invasive species, lessons, techniques.

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## INTRODUCTION

The response to the invasive alien species (IAS) problem has been increasing markedly over the last 10 years or so. An example of the increased profile is the classification of IAS as a "cross-cutting" issue in the implementation of the Convention of Biological Diversity, meaning that IAS will be considered in all other aspects of implementing the Convention and kept on the agenda for a protracted period. The raised awareness of the IAS problem has triggered a number of national, regional, and larger international projects and programmes. These have been instigated by various organisations and run by many different people from various backgrounds. These projects and programmes have been underway long enough to be able to conclude lessons about what works best and some of the technical needs that exist. In this paper, I have drawn on my own and colleagues' knowledge (obtained by interview and noted in the acknowledgements) and attempted to describe only projects or programmes that illustrate positive characteristics and lessons for the future. The great majority of this experience I have drawn on originates from work done in developing countries. Hence, this paper is not a comprehensive account of past and present IAS projects or programmes and it is entirely subjective. While there is a significant amount of other's information in this account, I take full responsibility for all that has been written. Most projects and programmes involve a number of different IAS, with vertebrate species just one amongst many, although I have not included information drawn from projects which specifically involve invasive plant species. However, this account is more about IAS generally, rather than specialising in invasive vertebrate species specifically.

## INTERNATIONAL INITIATIVES

The purpose of this section is to summarise some of what has been done at programmatic and institutional levels and include examples of some lessons have been learnt.

### The Secretariat for the Pacific Regional Environment Programme

The Secretariat for the Pacific Regional Environment Programme (SPREP) has had one of the longest running programmes on Invasive Alien Species in the developing world, having started in 1998. Originally the Programme Officer ran the IAS programme along with the Regional Avifauna Programme, which was logical because IAS are a significant threat to birds throughout the Pacific. The Programme has been funded by the New Zealand government (salary and some operating) and supported for project funding by the Australian and United States (US) governments. In recognition of the size of the IAS issue the SPREP created a full-time position for IAS and another for handling terrestrial species conservation on islands.

One of the main achievements of the SPREP IAS programme includes commissioning a review of the status of IAS (including vertebrates) throughout the Pacific. This formed the information basis to create a Regional IAS Strategy which was signed off by the SPREP Meeting at the formal meeting of the signatories to the SPREP and Apia Conventions (Sherley 2000). Other work included designing and writing a training programme for professionals involved in IAS as conservation officers or border protection officers, which has been carried out in five countries (Samoa, Vanuatu, Palau, Papua New Guinea and Niue) and two territories (American Samoa and Tokelau) in the Pacific. This programme formed the template for a similar training programme run by the Global Invasive Species Programme in parts of Africa and Asia. The SPREP IAS and Avifauna programmes also ran a number of threatened species programmes involving predator control or eradication. The idea was that the high profile threatened species (all birds) would help to profile invasive alien species. One of the programmes also developed a new "tropical formula" for a rat bait and experimentally determined minimum application rates to allow the recovery of the passerine kakerori, *Pomarea dimidiata* (Robertson et al. 1998, Robertson and Saul 2006). Finally, a programme called the Pacific Invasive Species Management Programme was designed and tendered to the Global Environment Facility (GEF). This programme reached the "PDF B" stage when it was "re-pipelined" for a second consideration. In effect, the programme is designed to start the implementation of the Regional IAS Strategy.

The structure of the proposed GEF programme could form the basis in part, or in full, for other funding proposals. The main components in the design of the bid are:

- Set up sub-regional centres of excellence for best practice, knowledge and training sites.
- Centres become "one stop shops" for sub-regional projects – supplying training in wildlife management including pest eradication and control. At least in the early stages they should be focussed on flagship species recovery programmes (including their habitat). Ultimately, these centres are run by islanders or locals for islanders/locals. Because these centres are sub-regionally based there is reasonable insurance of cultural compatibility.
- Centres supported by external expertise (such as the Global Invasive Species Database which receives some financial support for its services) whose objective is to pass on knowledge and skills (biological information on IAS, their control or eradication, expert lists, etc.) so that these can in turn be passed on by islanders/locals. Thus, technical information services ares not duplicated in the region.
- Identify sub-regional in-country projects supported by the regional centres; these projects capture an endangered species/habitat recovery programme (usually threatened, at least partly, by an invasive species) which serve as advocacy foci for the countries for the endangered species and invasive species prevention and other conservation messages ("flagship" species concept).
- The sub-regional centres are coordinated centrally by an inter-governmental organisation programme officer who coordinates external expert support (first stage training, etc.) and technical support such as technical information on the net (e.g., the Global Invasive Species Database).

Since the beginning of the SPREP-based Pacific region invasive alien species programme, others have been established, including the Pacific Invasives Learning Network (motivated by The Nature Conservancy) and the Pacific Invasives Initiative (funded by New Zealand AID). The Secretariat for the Pacific Community has also progressively worked more closely with the SPREP programme with its biosecurity programme. The Cooperative Islands Initiative has also started with its first phase, which is in fact the Pacific Invasive Initiative (NZAID-funded), which includes the above as partners and others (see below). This situation is ideal to allow cooperation and collaboration, probably best facilitated by the Programme Officer IAS (SPREP) serving to coordinate overall the implementation of the Regional IAS Strategy, which in turn, can do its part to implement the Action Strategy for Nature Conservation in the Pacific Islands Region (SPREP 2004). The latter could be seen as implementing the Island Biodiversity Programme of Work (motivated by decisions of the Conference of Parties to the Convention on Biological Diversity, see www.cbd.int/island).

Despite the apparent logic in this relationship, there are additional initiatives which need to be aligned with others. These include the National Biodiversity Strategy and Action Plans (NBSAPs). There needs to be some agreement between countries and major funding bodies regarding the relative roles of these strategies and plans and how these relate to funding agency and country priorities. The number of initiatives in the Pacific Region is causing confusion and diluting effort (and dollars) of the relatively few committed to conservation. One wonders how often this scenario of multiple "instruments," all doing similar things, has repeated itself around the world.

One of the roles of the SPREP has been to facilitate the creation and monitoring of the Action Strategy for Nature Conservation in the Pacific Islands Region (SPREP 2004) whose purpose is to serve as an over-arching document to guide conservation work in the Pacific. The most significant feature of the Action Strategy for Nature Conservation in the Pacific Islands Region is that the process for its creation was consultative over a wide range of representatives from all over the Pacific, representing government (including donor countries such as New Zealand and Australia) and non-government organisations (NGO) and other local representatives which met without political influence. The Action Strategy is now in its fourth iteration since 1977. The Strategy has been signed off by the SPREP, so it can be viewed as an official policy document. Monitoring the implementation (including designing indicators) is done by the Pacific Round Table which has a number of Working Groups, one of which is an invasive species working group. The Action Strategy pre-dates the CBD's Island Biodiversity Programme of Work, so their relationship is unclear. The Round Table reports every four to five years to a meeting of countries and non-government organisations which review the Action Strategy and make improvements to the strategy.

The latest report of the Round Table has shown that, although significant achievements have been made implementing the Action Strategy, it seems the original objectives were too ambitious and effort spread too thinly, leading to meagre progress against most declared objectives (including IAS). The operation of the Round Table has relied on voluntary contribution from members and has only formally a committed and funded part of a work programme for a few programme officers who have never been certain of long-term funding. On the other hand, there has been a wide variety of membership organisations and contributions have been without prejudice such as one might see in politically-motivated organisations. To date, the main benefits of the Round Table have been the opportunities for networking and planning collaborative projects. These benefits are difficult to quantify but, nonetheless, they are significant. To expect the Round Table to monitor the implementation of the Action Strategy for Nature Conservation in the Pacific Islands Region (SPREP 2004), including Invasive Alien Species, is probably unreasonable unless some major funding initiative arises to support it. The problems of the Pacific Round Table, in this respect, have been compounded (at least with respect to IAS) with objectives set in the Strategy which are far too ambitious and not designed with the technique of measuring performance in mind.

Over the last 10 years, the SPREP has allowed a number of key programmes which have been funded externally to be located in their buildings at Apia, Samoa. Physically locating these programmes, and often assuming the task of administering them, has allowed for a high degree of collaboration and cooperation between these programmes and those funded by SPREP such as the Regional Invasive Alien Species Programme.

**Pacific Invasives Learning Network**

One programme which has been co-located within SPREP in Apia is the Pacific Invasives Learning Network (PILN). The purpose of this programme is to create skills and knowledge bases and use these to seed further skills in others working on the same type of projects. There are eight partners in the Network: The Nature Conservancy, Secretariat of the Pacific Regional Environment Programme, IUCN Invasive Species Specialist Group, Conservation International, Secretariat of the Pacific Community, University of the South Pacific, National Palau Office of Environmental Response and Coordination, and the US Department of Agriculture's Forest Service. These partners, with help from the National Park of American Samoa, helped design and set up the PILN. In its early stages, funding has been from various sources, including The Nature Conservancy. There are six formed country teams (America Samoa, Guam, Niue, Palau, Pohnpei and Samoa) and six planned teams (Fiji, Hawaii, Kiribati, Kosrae, Marshall Islands and New Caledonia). Team members have various skills and all have commitments to conservation and invasive species management and prevention in their

countries. Country teams initiate in-country projects (such as developing a national invasive species strategy, public awareness programmes, eradication and restoration projects) and they are assisted by the PILN coordinator based at SPREP, who runs annual face-to-face and regular telephone conferences to enable sharing.

So far there are four distinguishing features that have materialised running the PILN: (1) the Network has generated further requests to join without any marketing, (2) six in-country teams were set up in May 2006 and a performance evaluation of all PILN individuals involved was conducted in November 2006, with 70% expressing high performance, 30% medium, and none low, (3) technical support for in-country projects has been done in such a way as to wean themselves from dependency from "outside" expertise so countries with a similar culture and technical issues support each other, and (4) the PILN has moved into marine invasive species as a direct consequence of demand from countries, starting with awareness raising. Thus, the PILN concept has accelerated faster than expected and seems to be a model which works and is popular. To date, the programme has been acutely short of funds, which may have been a blessing in disguise because it has forced the country members of the programme to include the projects and the necessary support into their existing work programmes (i.e., self-fund or find the funds themselves); this has forced countries to value and maximise ownership of the project. The concept should work in other regions and attract other investors because the start up funding leverages other sources of funds, including in-country sources. A future additional development is extending the concept into the agricultural and quarantine sectors, where need has been declared by the programme manager and countries.

**Pacific Invasives Initiative**

Another programme closely linked to the SPREP (as a partner) is the Pacific Invasives Initiative (PII). To date, the main *modus operandi* of the PII has been planning and implementing demonstration projects. These projects model best practice in planning and delivery, including community involvement. The programme has exacting standards for successful planning and implementation of invasive vertebrate species projects, including monitoring, project management, and community involvement. The concept for eradications includes placing all individuals of the target species at risk, ensuring that the method of removing the target individuals is faster than their intrinsic rate of increase, and ensuring that biosecurity of the island prevents further introductions. The success of the programme has been dependant on accessing technical support from New Zealand and networking in-country in the Pacific. It is often hard to determine, whether programmes should be situated in the Pacific where the eradication is needed or in a country like New Zealand near to the sources of expertise. The solution is to have the right partnerships of expertise with Pacific-based programmes and collaboration through all phases of a project (which the PII has been doing well). Funding bodies do not necessarily see having programmes based in New Zealand or "developing" countries as an ideal model, nor do programmes based in the Pacific necessarily identify with a programme situated "outside" the Pacific. The Pacific-based programmes can never (or at least not yet) have the immediacy of contact with expertise as those situated in centres of excellence, while the programmes outside need to have the experience of seeing a programme through to completion (in the sense of eradicating the pest species and securing biosecurity) to fully understand the range of needs in planning an in-country project.

**Critical Ecosystem Partnership Fund**

Another programme that uses "demonstration projects" is the Critical Ecosystem Partnership Fund (CEPF). This programme has been sponsored by Conservation International, the Global Environment Facility, the Government of Japan, the John D. and Catherine T. MacArthur Foundation, and the World Bank, who have created a fund which will fund projects focussed on biodiversity hotspots in developing countries. There are 33 hot spots identified around the world of which about 10 are already funded: Atlantic Forest (Brazil), Cape Floristic Region (South Africa), Choco-Darien-Western Ecuador (Columbia and Ecuador), Guinean Forests of West Africa (Cote d'Ivoire, Ghana, Guinea, Liberia, Sierra Leone and Togo), Madagascar and Indian Ocean Islands (Madagascar), Mesoamerica (Costa Rica, Nicaragua, Panama), Mountains of Southwest China, The Philippines, Sundaland (Indonesian island of Sumatra), and Tropical Andes (Bolivia and Peru). The CEPF supports projects run by non-government organisations and community groups but, apparently, not government agencies (including inter-governmental organisations). However, CEPF does make every effort to

coordinate its efforts with existing strategies and frameworks created by local, national, and regional government organisations (e.g., the Cooperative Islands Initiative in the case of the Polynesia-Melanesia Hotspot programmes).

Typically, the CEPF funds small- to medium-sized projects between 20,000-100,000 US dollars. In 2005, the CEPF received 1 million US dollars from the Australian Government's Regional Natural Heritage Programme for an IAS management programme. In one year, this programme funded 17 projects dealing with nine invasive species in the Polynesia-Micronesia Hotspot. Most of these budgets were under $20,000 but one, rat eradication on Alcipata Island offshore of Upolu, Samoa, received $140,000.

Later in 2007, the CEPF will launch a new 7 million US dollar fund for conservation in the Hot Spot, over a five year period, with about three million dollars being assigned to invasive alien species management. This programme is significant because the Ecosystem Profile (an investment strategy researched and negotiated for the hotspot) showed that IAS affected about 75% of the globally threatened terrestrial species. Habitat loss was the other cited cause of decline (Conservation International 2007). These causes of decline are probably aggravated by climate change and sea level rise. The hotspot spans 11 countries, 8 territories, and 1 US state (Hawaii) and covers a total sea and land area about 2.6 times the area of the continental US. Its land area is small, being 46,488 square kilometres (about the size of Switzerland) and includes about 4,500 islands (although 5 countries and territories are ineligible for receiving funding as they are not members of the World Bank and signatories to the Convention on Biological Diversity, Atherton 2004). Hence, in the 14 countries which are eligible, there are 161 sites with terrestrial threatened species (including turtles because they breed on land). Many of the globally threatened species at these sites are threatened by vertebrate invasive species such as rats (*Rattus* spp.).

**Global Invasive Species Programme**

The Global Invasive Species Programme (GISP) is a partnership between non-government organisations (The Nature Conservancy, TNC), government organisations (Government of the Republic of South Africa, Department of Internal Affairs; South African National Biodiversity Institute, SANBI), IGOs (CABI), the World Bank, and The World Conservation Union (IUCN). The latter is in itself actually a mixture of government and non-government organisations. From a business point of view, the last few years have involved contractual financial arrangements between the World Bank and the SANBI, with the former contributing dollars and the latter some dollars and considerable in-kind institutional support. The GISP also has a number of "Memoranda of Understanding" such as with Bionet International and national government agencies. Over the duration of its existence, the GISP has evolved, yet its prime purpose has been to facilitate the creation and implementation of IAS programmes around the world. In so doing, GISP put effect to Article 8(h) of the Convention on Biological Diversity which deals with IAS.

The GISP has: published advocacy and expert publications (including case-studies), run a sophisticated website which supplies and shares quality and up-to-date technical information, run regional workshops designed to produce IAS strategies, run specialist workshops (such as quantifying the economic cost of IAS to countries – especially with regard to poverty), published guidelines and best practice manuals for IAS management, designed and ran training workshops (initially in English and now in French and Spanish) for professionals in developing countries involved with terrestrial border biosecurity and post-border IAS management (including a course in prevention), run courses in the management of marine and coastal IAS (including ballast water), contributed to the Global Strategy on Plant Conservation (Target 10 Invasive Species) in particular identifying IAS indicators, and run special side-events at the CBD meetings and advised the Secretariat on implementing the COP decisions relating to IAS. IAS is considered a "cross-cutting" issue and, therefore, always on the agenda because it affects most other programmes.

The GISP has used a different approach to the SPREP in running its courses. Whereas SPREP has taken a country by country approach, the GISP has run courses for regions. These regions include southern and eastern Africa (2004 and 2005 introductory courses held in Kenya and Mozambique, 2005 marine and coastal course in Tanzania, and a 2006 course in prevention strategies in Tanzania), Black and Caspian Sea (2006 Istanbul marine and coastal course), North east Pacific and Caribbean Spanish-speaking countries (2006 Panama marine and coastal course in Spanish), and Francophone West Africa (2007 course on Marine and Coastal areas in French). It

is expected that a general/introductory course in IAS will continue to be run through the Global Environment Facility-funded African Barriers Project and is anticipated to include Ghana, Zambia, Uganda and Ethiopia. More training in legal aspects of IAS biosecurity is anticipated with World Bank funding for courses in Western and Eastern Africa and the South Pacific.

The GISP has also been involved in running the IAS sector of the Global Environment Facility's Biodiversity Indicators Programme. This work involves developing indicators for IAS and, amongst other things, reviewing relevant databases which might provide information on IAS indicators. Other GISP work includes: developing standards for the prevention of IAS via aircraft, planning contributions to the CBD COP 9 in-depth review of IAS, up-grading the GISP website (www.gisp.org) including developing a contacts database and electronic newsletter, a marine and coastal IAS brochure, and a booklet on IAS and poverty.

**Galapagos Islands Invasive Species Programme**

The Galapagos Islands Invasive Species Programme (GIISP) has already achieved some world firsts in invasive species eradications, such as eradicating goats (*Capra hircus*) off Isla Santiago (approximately 58,000 ha) and most of Isla Isabella (458,000 ha) as well as donkeys (*Equus asinus*) and pigs (*Sus scrofa*). The GIISP has also eradicated the little fire ant (*Wasmannia auropunctata*) from 58 ha Marchena (Causton et al. 2005). The scale of the operations involved in the IAS programmes in the Galapagos are extraordinary. For example, on Isla Santiago 66,329 goats were killed during 37 hours flying time and 59 forays. On Isla Isabella, 69,579 goats were killed in the 2004-05 season during 1,180 flying hours and 92,000 flying kilometres (F. Cruz, personal communication). Man-dog teams and Judas goats were used, especially when goat numbers were at their lowest. While these basic techniques have been known for years, the Galapagos team modified these to suit the extreme conditions on the archipelago. The pest animal programme is continuing to expand with the instigation of planning for the eradication of introduced rodents (there are still native rodents extant), starting with developing and ground testing techniques on relatively small islands with a view to moving to progressively larger islands as the skills (and funding) grow.

The concept behind the Galapagos Archipelago that is also extraordinary is that it attempts a "whole system" approach to managing IAS, including threatened terrestrial species and marine protection. This is based on a special law written for the protection of the Galapagos Archipelago, whose land area is approximately 97% in National Park status. The special legislation sets the framework for integrating all aspects of the administration of the islands, including land-use outside of the National Park, protected natural area and species management inside the Park (including National Park staff supported by a technical service provided by the Charles Darwin Foundation), quarantine, tourist management, etc. This concept of integrating all aspects of administering land (most especially integrating wildlife management with a technical support service) exemplifies a system that could be applied throughout the world.

**State of Hawaii**

Some in-country programmes are showing initiative with preventing and managing IAS. In Hawaii (US), the Invasive Species Early Detection Reporting Network aims to receive and act on information of new invasions in a timely manner to reduce the chances of a new IAS establishing. Early detection and eradication of a new invasion would minimise the ecological impacts of the species and the cost of eradicating it. Often, early intervention means the difference between being able to eradicate and IAS and having it become permanently established. The Network involves early detection, rapid assessment, and rapid response – all mediated through multi-agency (government) and private organisations or individuals. The Maui system includes a designated list of high priority species to watch for, outreach to train members of the public and professionals and semi-professionals working in natural resource management, user-friendly reporting and assessment system (e.g. by email, web-based system, walk-in drop off of specimens, see http://pbin.nbii.gov/invasives_report/online.asp), a searchable database to store early detection sightings, and reporting and action systems (also web-based and with 24 hour turn-around). Awareness raising is achieved with web-based identification fact sheets, a printed field guide, and workshops to provide a "search image" for the targeted IAS (e.g., www.reportapest.org), and simple methods for recording technical information. The fact that there is an easy to use system available encourages professionals, semi-professionals, and members of the public to actively search for invasive pests. So the existence

of the system itself encourages active searching by many people in various situations (termed by the authors of the system as the "eyes and ears network") which is more than can be achieved by agencies alone.

**Island Conservation**

Island Conservation (IC), like the Pacific Invasives Initiative, advocates technical and scientific excellence in their programmes. Programmes include island eradications from the Aleutians, British Columbia coast (Canada), Baja California, and Mexican islands off the Gulf of California. The projects all include systematic planning with set standards, which if they cannot be met, involve designing scientific research to support the planning objectives. Experts engage and consult with appropriate land owners, build ownership of the programme, plan, fund-raise, implement and set up monitoring of results. Implementation includes using the most modern methods and empowering local agencies. Pre- and post-operation monitoring of pest and beneficiary species is always done and written up for post-operation debriefing. Apparently, the IC has a policy of getting as many operations done as possible and, by iteration, improving techniques. IC exemplifies a business model for implementing its projects which, along with its pro-active pursuit of best practice and scientific basis of carrying out island eradications, functions as a target of excellence and standards for others in the international community to aspire to. Given this fact, it is important that the IC projects (how they were conducted, lessons learnt, etc.) are written up and disseminated to the wider conservation community. Technical transfer to local communities and agencies has been facilitated by forming sister organisations located in the areas where much of the historical work has taken place. These are the Conservacion de Islas in Mexico and Island Conservation Northwest in Canada.

An example of the type of project the IC runs has been on Santa Cruz Island (18,624 ha) in the Baja group of islands belonging to Mexico (one of about 23 eradication operations in the Baja Island Group). Other partners in the Santa Cruz operation included The Nature Conservancy and professional pest control companies (Pro Hunt and White Buffalo were hired to eradicate the pigs). Standard pre- and post-operation monitoring of target and non-target species was carried out. After the pig eradication, the feral turkey (*Meleagris gallopava*) population increased markedly and prompted an eradication programme for them. This eradication exemplifies how there can be unexpected outcomes from an eradication programme with the removal of the initial target species resulting in flow-on effects on other species. It also shows the importance of thorough and comprehensive monitoring, including non-target species.

**IAS Databases**

There have been a number of initiatives designed to provide technical support to operational programmes. One of the earliest of these is the Global Invasive Species Database (GISD). The purpose of the GISD is to provide high quality accurate information on alien invasive species, which is essential to running any IAS programme. In response to this need, the GISD was established in 1998 (www.issg.org/database). The GISD is run by two people out of Auckland University, New Zealand, as one of the programmes of the Invasive Species Specialist Group (ISSG). The ISSG is part of the Species Survival Commission, which in turn, is one of the commissions (like a programme) run by the IUCN. The GISD provides free information to conservation practitioners, decision makers and the general public. The basic unit of information in the GISD is a "species profile" (over 440 species) which provides information on how to identify the species (including diagnostic images), behavioural and ecological facts, and impacts and management methods including early incursion best practices. The GISD is also "networked" to other sources of information. These include experts who can advise on the species and links to other databases such as the IUCN Red List of threatened species, Ramsar sites where IAS are identified as threats, and the US Geological Survey's National Biological Information Infrastructure. The information is also available on CD ROM which is distributed to developing countries where internet access can be prohibitively expensive.

Testimony to the success of the GISD is in the frequency of it use – 1,100 unique visitors per day and a total of 75,000 hits per day. Feed back from users shows the database is being used for practical invasive species management, awareness raising activities (including education at schools), training, and assessments. Yet, despite the obvious demand for such a facility, the GISD has struggled for funding. The number of IAS databases as grown (two years ago I had found over sixty on the internet), resulting in duplication of effort in the area of IAS management. I have long advocated that IAS programmes include a component for

supporting the GISD as a sort of "tithe" so that it can add species profiles to its database. Over time, its size could increase and it could become the most comprehensive and authoritative source of technical information on IAS. With the diversity of information it contains, the GISD could become a "one stop shop" for practical programmes involving IAS.

The same principle applies to a new initiative called the Global Register of Invasive Species (GRIS). The ISSG in Auckland has developed a prototype database of the annotated names of known alien invasive species (meaning actually known to be invasive, potentially invasive, or posing disease risks somewhere in the world) which could be a standard tool to screen for and identify potentially risky organisms (e.g., pre-import screening of proposed imported species). The GRIS compiles and integrates invasive species names, which are linked to their threat status, location and data source annotations. The GRIS compiles and integrates lists of taxon names and associated information from multiple sources. All taxon names are linked to records of occurrence, native/alien status and invasiveness in specific geographic areas, along with associated information such as impact, spread or abundance. The GRIS will be able to capture (and donate to them) data from other sources of information such as the Global Biological Information Forum (GBIF) and the Global Invasive Species Information Network (GISIN). The GRIS database already contains 38,606 geographic records for 16,051 taxa, of which 1,453 species have records of invasiveness and 14,121 taxa are considered potentially invasive (where risk assessments have been done).

The GISD and the GRIS demonstrate the tremendous potential for data sharing on IAS provided by the World Wide Web. However, this full potential has not yet been realised because of the lack of commitment by an international agency to maintain a universal database. Thus, the leveraging and incremental benefits over time of having a centralised system are not yet realised.

## TECHNOLOGY AND METHODS

In this section, I am only going to give commentary on methods of pest eradication and control based on what I have observed with IAS programmes in the developing world and which exemplify improvements or best practice. Thus, it is not meant to be a comprehensive account. When it comes down to it, there are few generic methods available for pest animal control or eradication. Leaving aside biocontrol the same basic four methods are available today that have been available for thousands of years: trapping, poisoning, shooting, and hunting dogs (*Canis familiaris*, or other hunting animals). This makes us vulnerable to not meeting objectives when one of these tools is removed either through public objections or a tool becomes ineffective or unsuitable for technical reasons (such as large-scale operations). Hence, there is an imperative to start research into new methods now to remove this dependency on so few methods. Despite the small range of tools, we have not made as much of some as we could have.

For example, dogs can be trained to be efficient team-mates for hunters tracking down pest species, especially at low densities. Dogs have been used to great effect in this way in New Zealand and the Galapagos Islands. The latter programmes have used dogs in combination with Judas goats in the eradication of goats from Isla Santiago. Dogs (different individuals than those used for hunting pest species) have been used for searching for endangered species in New Zealand including such unlikely taxa as lizards, exemplifying the flexibility of dogs as tools. In New Zealand, the use of dogs has been structured so that there are systems for certifying dogs for a particular function, certifying the dog operators, and creating standards for the training programmes for the dogs and operators. These standards are necessary to avoid non-target species being killed and to ensure the most efficient use of man-dog teams is achieved. The necessity for standards of practice was exemplified in one reserve I visited where feral pigs were being eradicated using a mixture of techniques (e.g., man-dog hunting teams and trapping). In that situation, new dogs who had never hunted pigs before were being trained (without even the company of experienced dogs) on the target population of pigs. This meant that individual pigs that had escaped a hunt were in danger of becoming much more human-dog shy, and consequently, potentially very much harder to kill. Instead, the appropriate practice should have been to train the dogs on a non-target population of pigs until they were up to standard for use on the target population destined for eradication.

Some of the issues with using domestic dogs include quarantine if dogs are used between sites (especially between islands and countries), working out if particular breeds are more suitable than others, how to best maintain the dog's sensitivity to

detect the target species, how to best use dogs in combination with other methods such as aerial hunting, and continuing to develop better training methods.

In some situations, it may not be appropriate to aim for eradication using aerial broadcast of toxins or shooting from the air as parts of the tool box. This situation can result from inadequate biosecurity after the operation which may allow re-introduction. In developing countries with long traditions of customary use of land, land tenure disputes, and the inability to enforce biosecurity or quarantine standards, the biosecurity issue can often remove the option of eradication. The use of toxins may also be culturally unacceptable, or at least their protracted use. Despite this, pest animal control may still be required to retain threatened species or traditional horticultural practices. In New Zealand, improvements in trap and bait station technology, and aerial bait use separately or in combination, have greatly increased the possibilities for long-term control (not eradication) of pests over large areas. "Knock-down" operations using aerial 1080 (sodium monofluoroacetate) over areas as large as 40,000 ha have been achieved with tracking tunnel and residual trap catch indices of target pest species (possums [*Trichosurus vulpecula*] and stoats [*Mustela erminea*]) at or next to zero. Once large areas such as these have had the "knock-down" operation, "hold down" management can be implemented. This includes trapping, using custom-made kill traps which are designed to kill target mammals. Because the target species are at such low densities after the "knock-down" operation and the traps themselves are so efficient and target only one species, the time and effort to service them is minimal. Further, because the area is so large, the periphery to internal area is relatively small, the rate of reinvasion is reduced. Immigration of the target species back into the core protected area can be further reduced with more intensive control methods (perhaps including poison bait stations or man-dog hunting teams) focussed on the periphery. Further refinements on design can include focussing the "central" protected area on known distributions of key species threatened by the pest species under control. The benefits of the above scenario of protected natural area management include, using more acceptable methods to land owners (who, in developing countries, often occupy and harvest from the protected natural area) and employing the traditional land owners. Some of the research and management concepts which may be considered in planning such projects are discussed in Parkes et al. (2006) and Parkes and Murphy (2003).

Other technological developments suit themselves to low maintenance, minimum effort control programmes. Working versions of "control tunnels" have been developed in New Zealand which can sense which invasive pest species is entering the tunnel (e.g., rat or mustelid [*Mustela* spp.]) and deliver a dose of poison bait (e.g., cholecalciferol) to the animal as it passes into the tunnel or another machine made by another firm may apply toxin via an aerosol. Further modifications include small electronic cameras which can transmit an image of the animal which has passed through the tunnel, thus allowing a monitoring and research function (e.g., developing lures, baits and toxins) using this technology (see www.scentinel.co.nz, King et al. 2005). One of the great advantages of this technology is that the tunnels need not be serviced for very long periods of time (many months), depending on what functions they perform. Also, they necessarily mean that preset standards can be better met in field work. Their "per unit" cost is relatively high, but this may well be off-set if the alternative is another manual method which involves expensive man-power to service traps or poison bait stations. Of course, if these tunnels become widely used, their per unit cost should diminish.

Toxins are widely used and are often the only method available for invasive pest management or eradication. However, the widespread and prolonged use of toxins (especially via aerial methods) has its downsides. These include environmental risks, human health risks and animal welfare issues. Here, I relate from my personal experience as the leader of the applicants to New Zealand's Environmental Risk Management Agency requesting the assessment of sodium monofluoroacetate (commonly known as "1080") in order to validate the use of 1080 in terms of acceptable risks under New Zealand law (e.g., Hazardous Substances and New Organisms Act 1996). Over the last six years, the three main public bodies in New Zealand that use 1080 applied to the Environment Risk Management Authority (Department of Conservation 2006) to have it formally assessed for its public health and environmental safety under relevant statute criteria. The process was long and drawn out and included a thorough technical and scientific assessment of its safety, along with a public submission and hearing phase. The process was extremely exacting,

exposing the use of 1080 to the closest scrutiny imaginable. It drew on all facets of technical information (scientifically defensible and otherwise) and the public's views of the use of a broad spectrum toxin and its aerial application.

In New Zealand, 1080 is used on the mainland as a toxin to control introduced pest animals which threaten native biodiversity, one of which is also a vector for bovine tuberculosis (TB) whose prevalence threatens exports of cattle products. The species targeted for the latter is the Australian brush-tail possum (*Trichosurus vulpecula*). During these control programmes other species which threaten native biodiversity are killed incidentally, but effectively, including rat species and mustelids. The toxin, 1080, is one of only a few "toxin tools" available to conservation and TB managers for effective management of pest species. Other methods are being investigated (such as biocontrol and technological methods), but these are likely to be years, even decades away from being available for widespread use (see below). The use of 1080 in conjunction with other toxins (principally brodifacoum, cholecalciferol and cyanide), involves bait stations or aerial methods (mainly using helicopters with under-slung buckets with mechanical "spreaders" with spinning blades to throw out the baits as the machine flies over a pre-set course using differential geospatial positioning systems). In the early 1970s, 1080 carrot baits used to be used aerially broadcast at up to 30 kg/ha in native forests to control possums, resulting in reductions of only about 65%, based on residual trap catch indice, RTCI, figures, whereas today application rates are down to less than 3 kg/ha with RTCI's of over 95% (Department of Conservation 2006). The successful "by-kill" of other invasive species such as mustelids and rats can also be now relied upon using 1080 (Brown and Urlich 2005).

The benefits to native species of using 1080 has been supported by extensive scientific literature (New Zealand Environmental Risk Management Agency 2007). However, the assessment brought to light technical and "social" short-comings in the use of broad-spectrum toxins in wild-lands and the attitudes of communities. The volume of scientific literature on the effects of 1080 on target and non-target species and social aspects is very large (New Zealand Environmental Risk Management Agency 2007), and the checks and balances on the handling and use of the toxin are exhaustive and continually being improved. Over the forty plus years of using 1080 in New Zealand, the publicity (mainly bad) has meant that there has been unparalleled public debate over its safety and efficacy. Despite the history of research and debate, there were some clear gaps in knowledge and lessons about public perceptions on the use of toxins, especially using aerial application methods. The public distrusted expert scientific evidence, even if it was based on extensive peer reviewed research and literature. Scientific evidence was insufficient to answer cultural and spiritual objections to the use of a toxin in natural environments and there was an obsessive concern about toxin getting into water and flow-on effects to the ecosystem, generally or risks to non-target species including humans. Despite evidence to the contrary, there was a reluctance to believe that there were benefits to native biota. Rather, people clung to believing the opposite, citing as evidence their (qualitative) observations of declining number of animals after the use of the toxin, the effects of long-term use of the toxin (and the by-products of its breakdown) on the ecosystem and people even at extremely low concentrations, and the humaneness of the manner of death of target and non-target species was questioned.

Given the few tools for invasive vertebrate pest control and eradication, there is a premium on finding new methods. One group of possibilities that are being investigated in New Zealand have been termed "biotechnological" which includes some aspects of classic biocontrol. These methods are aimed at possum control and have potential for other mammals, although there are some fundamental differences in physiology between marsupials and eutherian mammals. There are three basic avenues of enquiry: Zona Pellucida (ZP) protein-based vaccines (including using virus and a host-specific nematode vectors) which prevent fertilisation, hormone toxin conjugates (inducing permanent sterility with one dose or hit by targeting the pituitary gland and GnRH), and possum gut ion transporter system toxins (see www.possumbiocontrol.agresearch.co.nz). The three million New Zealand dollars per year programme is multi-disciplinary and involves New Zealand and offshore institutions working in a complicated, integrated manner. The programmes will be funded for four years by which time proof-of-concept for at least one of the techniques must be reached. After this time, a further eight years research will be triggered to continue the successful option towards the point where it can be tried in the field.

Some of the issues which have emerged so far include the concern that any vaccine delivery (of hormonal toxins) involving a bait will still attract

the same objections currently aimed at baits used for delivering poisons; the practicality and desirability of distributing baits with vaccines over the vast areas which will be required for widespread control; public mistrust of biotechnologically originated methods; the depth of research needed (including "basic"/first principles physiological research) and, hence, time required to bring the methods being pursued to proof of concept stage. The latter is perhaps the most significant if indeed it is indicative of this general area of research because of the sheer volume of research that has to be done to develop the new methods to proof of concept stage, and the concurrent cost and time to bring the method to a field delivery standard. Current reckoning is that the time yet to develop a field deliverable method is possibly decades away.

## REQUIRED IMPROVEMENTS

### Technical

Many of the gaps in methods and techniques have been described. The needs for development are driven by societal and target and non-target species considerations. There is also at least one "commercial" one as well. The problem is that the size of the "market" for using new technology is often too small in any one country to justify the research and development investment required to create the new technology. Thus, it is time that international programmes or agencies pool their resources (possibly along with the private sector) to create the economies of scale required to afford the investment needed to create new methods for controlling, eradicating, and monitoring pest species. This cooperative approach should win favour with funding organisations which typically look for integrated, "leveraged," and cooperative projects. Funding agencies will have to be educated into the long-term nature of research and development of new techniques. A multi-disciplinary team approach is needed with focussed teams of sufficient number to get past that critical mass required to achieve results in a reasonable time frame.

New techniques that are needed include:

- Improved methods of using existing toxins such as 1080 by developing new bait formulations (including attractants for target species and repellants for non-target species) and applying these toxins at strategic periods during the annual cycle of the target species such as when food supply or other environmental stresses (e.g., drought) are at their maximum. Improvements have been demonstrated in the past with the use of 1080 in New Zealand with the application rate in forested areas in the 1970s at about 30 kg per hectare and achieving about 70% kill of the target species (brush-tailed possum) compared with the current application of less than 3 kg per hectare and reductions of at least 90%.
- New toxins which are as species specific as possible in terms of their target species toxicity or in the way they are deployed (bait stations or bait design).
- Methods for using toxins in combination with each other using different methods of application and trapping in different habitats.
- Monitoring methods for target species, especially at extremely low densities.
- Research which demonstrates the benefits (e.g., to threatened native species) of long-term periodic (pulsed) pest control, especially over large areas and efficient temporal and spatial designs of establishing large areas in which pests are initially reduced with a broad scale application method (such as aerial application), then "protected" by minimising immigration of target species (by setting up barricades surrounding the protected area using traps and/or poison bait stations) and reducing recovery of surviving target species within the controlled area, all balanced against the requirements for beneficiary species to recover as viable populations or to re-establish them. Such management of large protected areas surrounded by a "barricade of protection" also requires new technological tools such as monitoring pest and beneficiary species at low densities.
- Data which will assist showing the benefits of removing pest species need to be collected over long periods of time. Short-term benefits are relatively well documented. The recent experience of the 1080 reassessment in New Zealand has been that conclusions based on long-term monitoring research are required to mollify the critics of using toxins and these studies are almost non-existent. It needs to be kept in mind that the developing countries deserve the same standards and considerations as the developed countries for using toxins. This is particularly relevant for many of the island programmes now

underway which will in the future involve, more and more, inhabited islands. Public education on the use of invasive pest control benefits (especially toxins and their aerial application) need to be done in the context of explaining the benefits to indigenous biodiversity, agriculture or human health so that any perceived down-sides are considered in the context of benefits.

- New technology such as biocontrol and biotechnological methods that will, in the long-term, remove our dependence on the few "traditional" methods (poisons, trapping, shooting, and hunting dogs) need to be researched. However, we need to recognise that the time frame for these avenues of research to generate methods available for use in the field are many years (perhaps decades) away, judging from our experience with possum biocontrol and biotechnological control in New Zealand. This is because the research required is quite fundamental biological research before it can be progressed to the "applied" stage and there are major checks and balances relating to human and environmental safety which must be overcome. Also, the scientists involved in this research are at pains to point out that no method being developed is likely to ever be a "silver bullet" which will singularly reduce invasive pest numbers down to target levels. Therefore, in the short to medium term (possibly even long-term if you consider that it is unlikely that a given new technology is ever likely to be the complete "silver bullet" for pest control or eradication) we need innovative methods of delivering toxins and trapping so that only selected species are targeted in the most efficient, cost-effective manner possible. Again, the best approach is to take a collective, cooperative approach in developing these methods to achieve economies of scale and complete development of new methods in an acceptable time frame.

In the long-term, the need to deliver a control technique over a large scale will always be the limiting factor requiring redress because much of the land which must be worked is rough and inaccessible or the available man-power is too limited or costly to enable using manual methods such as poison bait stations, hunting dogs, or trapping. Therefore, long-term we need to develop more species specific toxins and bio-control and biotechnological methods, especially as the size of areas requiring eradication or control get progressively larger.

**Programmes**

Many of the international programmes (such as the GISP) have promoted and facilitated the creation of various "strategies" or plans. While this may appear laudable, it is not without its downsides. Many of the "strategies" have been developed in large and expensive workshops which have taken a lot of preparatory work and follow-up with publication of the proceedings. These workshops have been run without any planned funding in place for their implementation. Thus, the worst outcome has happened in the past, that of raising expectations in-country and not meeting them. This has developed cynicism and mistrust of "outside" developing nations and their initiatives. Funding agencies supporting these types of workshops must be signed up for follow-up funding arrangements to implement work-plans arising out of the workshop. On the other hand these workshops do act as advocacy for work which might attract funding agencies (especially if there is formal endorsement by countries or other "official" bodies) and, thus, serve as leverage or initiate programmes.

One of the fundamental functions of programmes such as the GISP has been to coordinate effort around the world or regions to avoid duplication of effort and to combine effort or facilitate the creation of complimentary programmes. Despite this, much duplication is occurring such as the creation of databases on IAS, including various aspects. Part of the problem is that the GISP and other regional programmes (such as the SPREP) are not formally able to direct agencies or programmes towards any particular activity and agencies or programmes are not formally required to engage the GISP or other programmes and to seek and take advice. Further, over-arching organisations such as the CBD have harboured the rather naive view that programmes such as the GISP can somehow effect the required coordination. This inability is especially acute if those same agencies trying to achieve coordination are strapped for cash. Without cash to fund the coordinated type of projects required, they have very little influence. Further, these high-level programmes are struggling to find sufficient funds to keep solvent themselves. Thus, they are spending a lot of time "surviving" (acting like

consultants undertaking projects) rather than working in a role as coordinators and facilitators.

The issue of "working to role" of programmes with large mandates such as the GISP (but also the SPREP IAS programme) is in itself often a point of contention between colleagues. Many think that these programmes should not be actually doing project work, but rather this should be the left to countries, NGO's, and the like ("smaller" players). Instead, it is perceived that the programmes such as the GISP should be focussed on generic international-level work such as policy, coordinating and facilitating projects and finance, etc. While this may be the ideal, funding is very difficult to find for "secretariats" doing this generic-level work. The reality is that funds can be far more easily won by bidding for project support.

Some programmes have as one of their maxims technical excellence and using the highest technical standards to establish projects involving eradication. These standards involve well known practices which include placing all individuals of the target species at risk, killing target individuals at a rate that exceeds the rate of reproduction and immigration, and ensuring the future biosecurity of the site or island after the eradication. These and other "rules" are absolutes if true eradication is to be achieved and the failure to guarantee them (especially the latter on biosecurity) has caused debate amongst colleagues and the postponement of operations. This desire to exact such high standards has been sharpened because these projects (mainly in developing countries) may be "demonstration projects" one of whose objectives is to showcase best management practices without compromise. Thus, in effect, the implementation of at least one project has been stalled because, ostensibly, the necessary standards for an eradication operation have not been met. However, could this risk-averse approach be preventing progress? If one reflects on the lessons learnt from the developed countries, much of the "know-how" has been generated by making mistakes (or undertaking some operations "the hard way") and iteratively building up knowledge. While it is obviously beneficial to learn from other's experience, in order to embed knowledge securely in a new culture, I think there is also a need to make provision for "learning by experience" in the developing world which might mean not doing things in the best possible way every time. Thus, there needs to be some risks taken, where some of the accepted prerequisites (or standards) for eradication (or control operations) may not be met so that there are provisions for learning and adapting as the programme progresses. The project might be allowed to start without all the exactitudes of every standard met *apriori*. This recognition of the requirement to learn while implementing demonstration projects is further justified by the fact that, almost certainly, not all practices learnt from the developed world will apply the same way in the developing world. For example, many monitoring and bait deployment practices have been developed in temperate countries and will not work as well in tropical countries because of new types of non-target species such as land crabs which interfere with equipment and consume large amounts of baits.

Demonstration projects are commonly proposed in funding bids for programmes such as the Pacific Invasives Initiative and the Critical Protected Ecosystems Fund. As discussed, the concept is to provide a learning model for local implementing agencies to acquire the skills necessary to carry out further projects. Most demonstration projects are selected on the basis of biological characteristics such as the presence of threatened fauna and flora. Yet, if capacity building is in fact the main objective then more time needs to be spent on selecting local implementing agencies which can be involved in a series of projects. These projects should gradually place progressively more responsibility on the local implementing agency for instigating, planning and implementing the project. In this scenario, the "external" support would shift from one of being planner, designer and teacher to one of finally providing peer review and advice only. Thus, it may be more important to place emphasis on selecting the right implementing agency rather than putting so much emphasis on the biological qualities of the projects.

Another feature of the CEPF is its policy to support local civil organisations such at the village or NGO level and with relatively small grants. Thus, the demonstration projects which seem to be feasible under this regime would appear to be one-off projects. This policy might be improved if the Fund followed the principle of empowering such agencies with a series of projects which progressively built the capability required to finally independently design and run projects. Thus, multiple projects involving single agencies responsible for fewer locations funded by larger grants maybe a better model to follow. It would also be advisable to involve relevant government (and possibly IGOs) in the process of running the projects to maximise the leveraging of support and

skills (e.g., networking with Secretariat of the Pacific Regional Environment Programme [an IGO] and the Pacific Invasives Learning Network [a NGO]). The CEPF funded projects may also be a prime opportunity to integrate with the Cooperative Islands Initiative (CII) launched at the Convention on Biological Diversity's 6th Conference of Parties at The Hague, 2002, which is a global partnership concept involving NGO and IGO agencies cooperating on a collective work programme. Indeed, the Pacific Invasives Initiative (itself the first initiative under the CII) has been suggested as assisting the establishment of the demonstration projects described in the Micronesian Polynesian Hotspot, CEPF programme (thus incidentally, involving IGO's in the CEPF programme after all because the SPREP, an IGO, is a partner of the PII).

In a similar vein, there is further networking possible for the CEPF and its partners with the Global Island Partnership (GLISPA) which was launched at the CBD COP8 in Curitiba in March 2006 by the President of Palau and Indonesia, and others. The GLISPA was launched with the "Micronesian Challenge" which includes committing 30% of marine and 20% of remaining forests to becoming protected natural areas. The "challenge" has been to other countries and territories to match it. The purpose of the GLISPA is much wider though because it hopes to implement the Convention on Biological Diversity's Island Biodiversity Programme of Work. Hence, there are plenty of initiatives all with the mandate to promote IAS programmes. The challenge will be for countries to settle on one to provide leadership (and hence, coordination and collaboration) which will require significant investment.

While early detection and intervention is not, strictly speaking, preventing the introduction of new species, systems as established on Maui in the Hawaiian Islands are ideal, provided the community has the technology, because the system potentially mobilises the whole community to keep watch and provides unity of purpose and shares the risk posed by IAS. The concept of the IAS Early Detection Reporting Network as set up in Maui, Hawaii, may not be as workable in some developing countries. However, the principle is excellent and, indeed, will serve a number of functions including the obvious of early intervention and eradicating potentially invasive alien species before they establish, and raising the profile of IAS in the minds of the public and transferring some of the responsibility for their management on to the public. As with many IAS initiatives, the real benefits of programmes such as these will only be realised with long-term funding. The point being, it is essential that a reasonable time period is included in planning because the input of the community (which is so critical) is essentially about changing behaviour which takes a long time.

Of all the options available for IAS work, prevention still must rank highest priority. The problem with prevention work is that zero occurrence is the measure of success. Thus, by definition, there are no tangible outcomes for a successful prevention programme – not an attractive scenario for a funding agency which wants to see material outputs and outcomes. The international conservation community needs to make as much as possible out of the consequences of the break-down of biosecurity measures when they happen (such as occurs with invasive rat invasions) and drive this point home to funding agencies so that the value of prevention programmes can be placed in perspective.

## SYNTHESIS AND RECOMMENDATIONS

Based on my experience and the commentary from colleagues, I offer the following analysis. Considering all the money and effort that is being spent on IAS programmes in the developing world (and this would be true of at least others addressing threatened species conservation), the ratio of planning for, advising and investigating the problem to actually doing something about it on the ground is too much weighted towards the former. Too often, analyses of problems (like IAS issues), planning and "advising" is done without the follow through of completing the projects on the ground. Thus, the expectations of in-country people are raised without these being met and cynicism sets in.

In considering what projects should be attempted, more attention should be given to empowering selected institutions aimed towards them acquiring progressively more and more capacity to be able to plan and run projects themselves. This scenario is more preferable than trying to address a wider range of biological topics. Thus, weighting projects this way should better address the issue of permanently empowering local institutions with the capacity to meet their conservation and IAS needs.

In this paper, I have mentioned only a small number of the international programmes involved

in invasive alien (vertebrate and other) species. Programmes relating to island conservation alone include the Pacific Invasives Initiative, Cooperative Islands Initiative, Island Biodiversity Programme of Work (under the Convention on Biological Diversity), Global Islands Partnership (partly set up to implement the IBPOW), the Critical Ecosystems Partnership Fund, Island Conservation, and so on. There are also numerous database-style programmes involving invasive alien species (e.g., the Global Invasive Species Database, Global Invasive Species International Network) and other programmes or projects which support on the ground work. These programmes include many sub-programmes and projects which multiply the complexity of the situation. With so many programmes and projects working on invasive alien vertebrate (and other) species, there must be a lot of duplication of effort. A lot of work needs to be done between programme leaders to network and share expertise. Programmes such as the Global Invasive Species Programme are mandated to do this yet they struggle to win funds to play this role in the global community. The task is huge yet, after about 10 years in existence the GISP is still tiny and cannot cope with the task at hand. The role of networking and sharing expertise and knowledge seems unattractive in itself to funding agencies. Most want to see more tangible output based projects in their portfolios. For their part, programmes such as the GISP need to work on methods to quantify how much added value they bring to other programmes and projects. Thus, there needs to be a turn-around in attitude of funding agencies which will mean funding "secretariat" style programmes such as the GISP and recognise their "added-value" benefits to other "on the ground" projects. One solution could be to follow Island Conservation's model of creating "sister organisations" in the regions they work in and programme the costs of establishing these into the project costs of the island eradications in the region. In this way, locals are being empowered to carry out their own conservation work and the model of the implementing agency (like IC) moves progressively from demonstration projects to being advisors to local agencies carrying out the conservation project.

Funding agencies and planners need to consider how many similar projects there are in existence today and whether new ones are really necessary, or whether it would be better to invest more in existing projects and programmes. There is certainly duplication in purpose with various IAS projects and in the "service" programmes such as the information databases. Thus, I think it is time to seriously think about a consolidation phase in funding new work under existing programmes umbrellas rather than starting new ones. Related to this issue is the need for projects and programmes to more efficiently trade information for obvious reasons. This improvement may be achieved by funding bodies insisting on making this networking happen in the planning and implementing phases of projects and programmes. Some of this trading of knowledge and skills can (and should) be the responsibility of some key organisations such as the Global Invasive Species Programme and other secretariat-style programmes. In turn, these should be valued by funding bodies as necessary for helping to ensure "operational" programmes are as cost-effective as possible, receiving as much benefit from other projects as possible. This "added value" capacity of "secretariat" like programmes should be seen as sufficient justification for "secretariats" being funded.

Non-government organisations need to retain their link with the government sector because, in the long term, the government agencies need to be up-skilled if some continuity of improvements is to be achieved. Non-government organisations can and do provide much of the practical, on-the-ground work at least in the Pacific and, in so doing, provide a stark reminder to government agencies of what they should be themselves doing. While this is important, nevertheless, NGO's should make every effort to work in with government agencies.

One of the most difficult problems to overcome is long-term funding (meaning decades) to ensure that a project will continue. Funding time-frames are typically one to five years – too short to ensure that local capacity has been improved sufficiently to ensure the continuity of the programme. This is particularly significant as a problem because the biological and ecological nature of the issues coupled with the changes in human behaviour required mean that significant periods of time are required to effect long-term changes. One aspect of this need for long-term planning is for on-going biosecurity after an eradication has been done. The question of continuing biosecurity to prevent re-invasion has been the stumbling block for deciding whether a number of projects should go ahead. The basic problem is the time frame for the eradication operation itself and ensuring ongoing biosecurity are completely different (the latter being much longer) and are also different in the nature of the work required. While some funding and

implementing agencies are aware of the biosecurity issue, others apparently are not. The above also needs to be considered in the context of not making project and programme objectives over-ambitious.

The future for making rapid enough progress in technical improvements lies with international cooperation. Individual agencies cannot cope with improving trapping, monitoring, and toxin-based methods by themselves because the economies of scale will not allow it. For conservation purposes, it is unlikely that commercial corporations will be attracted to researching and developing new methods because the market is too small. Conservation agencies and funding organisations need to consider pooling resources on common technical problems. I think there are enough similarities of issues between countries to make this feasible such as the need to eradicate rodents off large islands. Labour saving technical advances are possible in such labour intensive activities as tracking tunnels, species specific traps, automated traps, and monitoring devices which traditionally take up huge components of the total budget of the project. These labour-saving improvements will allow (1) staff to be redirected to other conservation work such as threatened species management or biosecurity, and (2) much larger areas to be managed (control or eradication) than possible at present.

The rigorous exercise of registering 1080 in New Zealand as a "safe" toxin from environmental and human health points of view highlighted to me the biggest weaknesses of using broad-spectrum toxins that can kill a number of different species apart from the target species. These include the increasing opposition (especially from traditional cultures) of killing non-target species and the objection to "spiritually corrupting" the ecosystem by applying toxins, especially using aerial techniques which, of course, are the very methods we need to use for larger areas. Another problem with acute and chronic toxins (such as 1080 and brodifacoum, respectively) is the manner of death of the target and non-target species (especially where the latter are large-bodied animals which may only get exposed to sub-lethal doses) because of animal welfare issues. For 1080 and brodifacoum, there is insufficient known about the manner of death of target pest species and what is known of the manner of death of wild animals seems to be inferred from research based on captive or laboratory animals. Thus, there is little known about the pathology of toxins in wild animals (see review papers such as Mason and Littin 2003, Littin and Mellor 2005). Research is needed on the pathology of these and other toxins commonly used on target and non-target species known to take toxins to off-set the inevitable opposition likely to increase from the animal rights movement.

## CONCLUSION

In conclusion, I think there are some well known lessons about how best to design invasive alien species programmes or projects in "developed" and "developing" countries alike and the real challenge is to heed these lessons in planning and implementing them. There is an obligation on funding organisations and those proposing programmes or projects to jointly avoid duplication of effort, and plan and commit to long-term support to maximise the chances of transferring capacity to local organisations. Local capacity building, the focus of much NGO work, needs to be coupled with "national" (often government agencies) agencies to help ensure longevity of the transfer of capacity. Our dependency on the few control and eradication methods we have available needs to be reduced. Hence, new methods of killing (such as p-aminopropiophenone – PAPP) and monitoring pest, and some particular taxa of non-target, species are needed which are species specific. This research and development is urgent and needs to be tackled cooperatively between international organisations. Finally, as has been repeated elsewhere many times, prevention must be the priority and new methods (such being developed in Maui, Hawaii) for ensuring biosecurity are essential.

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## LITERATURE CITED

ATHERTON, J. 2004. Critical ecosystem partnership fund ecosystem profile polynesia and micronesia hotspot. Conservation International Melanesia Center for Biodiversity Conservation. Apia, Samoa.

BROWN, K. P. AND S. C. URLICH. 2005. Aerial 1080 operations to maximize biodiversity protection. New Zealand Department of Conservation Research Series no. 216. New Zealand Department of Conservation.

CAUSTON, C. E., C. SEVILLA, AND S. D. PORTER. 2005. Eradication of the little fire ant, *Wasmannia auropunctata* from Marchena Island, Galápagos: on the edge of success? Florida Entomologist 88:159-168

CONSERVATION INTERNATIONAL. 2007. Critical ecosystems partnership fund fact sheet, www.cepf.net.

DEPARTMENT OF CONSERVATION. 2006. Application for assessment of sodium mono-fluoro-acetate (1080) under the Hazardous Substances and New Organisms Act 1996. Department of Conservation, Wellington, New Zealand, www.ermanz.govt.nz, 1080 reassessment.

KING, C. M., R. D. MARTIN, R. M. MCDONALD, G. TEMPERO, A. DEKROUT, S. HOLMES, AND M. STIRNEMANN. 2005. Automated monitoring for detection of small mammals. Pages 106-107 *in* 13th Australasian Vertebrate Pest Conference, Wellington, New Zealand.

LITTIN, K. E. AND D. J. MELLOR. 2005. Strategic animal welfare issues: ethical and animal welfare issues arising from the killing of wildlife for disease control and environmental reasons. Review of Science and Technical International Epizootics 24:767-782

MASON, G. AND K. E. LITTIN. 2003. The humaneness of rodent pest control. Animal Welfare 12:1-37

NEW ZEALAND ENVIRONMENTAL RISK MANAGEMENT AGENCY. 2007. Evaluation and review report. Application for the reassessment of a hazardous substance under Section 63 of the Hazardous Substances and New Organisms Act 1996. Name of substance sodium fluoroacetate (1080) and substances containing 1080. Environmental Risk Management Agency New Zealand. (www.ermanz.govt.nz and go to 1080 reassessment).

PARKES, J. AND E. MURPHY. 2003: Management of introduced mammals in New Zealand. New Zealand Journal of Zoology 30:335-359.

PARKES, J. P., A. ROBLEY, D. M. FORSYTH, AND D. CHOQUENOT. 2006. Adaptive management experiments in vertebrate pest control in New Zealand and Australia. Wildlife Society Bulletin 34:229-236.

ROBERTSON, H. A., E. K. SAUL, AND A. TIRAA. 1998. Rat control in Rarotonga: some lessons learnt for mainland islands in New Zealand. Ecological Management 6:1-12.

ROBERTSON, H. A. AND E. K. SAUL. 2006. Conservation of kakerori (*Pomarea dimidiate*) in the Cook Islands in 2004/05. New Zealand Department of Conservation Research and Development Series 246. New Zealand Department of Conservation, Wellington, New Zealand.

SHERLEY, G. H. 2000. Invasive species in the Pacific: a technical review and draft action strategy. South Pacific Regional Environment Programme, Apia, Samoa..

SPREP. 2004. Action strategy for nature conservation in the Pacific islands region: 2003-2007. South Pacific Regional Environment Programme, Apia, Samoa.

# A "COMMUNITY" APPROACH TO INVASIVE SPECIES MANAGEMENT: SOME PACIFIC CASE STUDIES

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***Abstract:*** Conservation is essentially a social activity – it is about, by and for people. Managing invasive species on islands to reduce their "community effects" (that is, effects on communities of people as well as on communities of plants and animals) makes good sense. Consequently, many demonstration projects supported as part of the Pacific Invasives Initiative involve social and economic objectives as well as biological ones. In New Caledonia, for example, dialogue between conservation agencies and local tribes about a proposed pest mammal control project on Mont Panié has been taking place for nearly 10 years. As well as being involved in pest control activities to achieve biodiversity conservation outcomes, there is particularly strong local support for a proposed project to reduce feral pig (*Sus scrofa*) impacts in tribal gardens. It is anticipated that reduced pig impacts will lead to important improvements in peoples' livelihoods. A proposal to eradicate invasive cane toads (*Bufo marinus*) from Viwa Island in Fiji to protect an endangered ground frog was modified following consultation with local residents to first include the eradication of feral cats (*Felis catus*), feral dogs (*Canis familiaris*), and rats (*Rattus* spp.). These preliminary eradications were undertaken by the residents, with training and support from eradication specialists. Viwa islanders now have the knowledge and skills to ensure their island remains free of these pests. Also, Fiji Vatu-I-Ra Island is an important seabird breeding site. Local residents gave their support for rats to be eradicated from the island. Several people were trained and participated in the eradication operation. In addition to removing predation pressure on nesting seabirds and creating opportunities for an eco-tourism business, another early outcome has been increased measures by the owners to protect the island from further human-induced impacts. These and other demonstration projects are discussed to illustrate attempts at achieving biological, social and economic objectives through managing invasive species on islands.

***Key Words:*** community approach, conservation, invasive species, island conservation.

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## INTRODUCTION

Conservation is essentially a social activity – it is about, by and for people. Managing invasive species on islands to reduce their "community effects" (that is, effects on communities of people as well as on communities of plants and animals) makes good sense. Consequently many demonstration projects supported as part of the Pacific Invasives Initiative (PII) involve social and economic objectives as well as biological ones. We discuss three PII-supported demonstration projects where socio-economic objectives have been progressed alongside biodiversity conservation ones.

## PEST MAMMAL CONTROL, MONT PANIÉ, NEW CALEDONIA

With a unique and diverse biota, New Caledonia is ranked as one of the world's "Biodiversity Hotspots". Many native plants and animals are endemic, and threatened, by human-induced changes including nickel mining, fires, and invasive species. Relationships between New Zealand and New Caledonian agencies have developed in recent years focused on biological similarities between the two Gondwana remnants, and on common approaches to addressing threats to biodiversity in both countries. In the late 1980s, the Maruia Trust coordinated a rapid biodiversity survey at five sites in New Caledonia. In consultation with the Government of Province Nord, it was subsequently decided to develop a conservation management

project at one of these sites, the Mont Panié Special Botanical Reserve in the north-east of Grand Terre. Mont Panié is the highest mountain in New Caledonia. Its flanks are covered with the largest tract of rainforest remaining in the country. With the added support of Conservation International, the 'Mont Panié co-management conservation project' was initiated in 2003.

While there is no nickel mining in the northeast, a combination of fires and a range of invasive plants and animals are leading to significant biological degradation in the Mont Panié area. Important progress has already been made in raising awareness among local people of the impact of fires on biodiversity values, and on peoples' livelihoods. The number of fires being deliberately lit is declining in some areas. Community awareness of invasive species issues has also been raised. Dialogue between conservation agencies and local tribes about proposed pest mammal control on Mont Panié has been taking place for nearly 10 years. In 2003, a visit was arranged to New Zealand (NZ) for a group of local people from Mont Panié so that they could see pest mammal control projects first-hand, and to talk to local stakeholders. In line with an aim to involve and train local people, the Association Dayu Biik was established which includes representatives of Kanak tribes and clans living on the flanks of Mont Panié. The main purpose of the Association is to provide a conduit for communication between the tribes and the conservation project and to ensure local perspectives are properly reflected in decision-making.

Because controlling invasive species typically involves intensive and on-going actions, an important aim of the co-management project is to develop local capacity to achieve pest control targets and to sustain desired outcomes. Initially, six pest mammals have been targeted within the Mont Panié reserve: two species of rat (*Rattus* spp.), cats (*Felis catus*), dogs (*Canis familiaris*), deer (*Cervus* spp.), and pigs (*Sus scrofa*). In addition to causing biodiversity declines, some of these mammals also have economic impacts. The NZ Department of Conservation (DOC) was initially engaged by the project partners to assess the feasibility of these mammals being controlled and of outcomes being sustained by local people. Following its establishment in 2004, the PII facilitated and coordinated specialist inputs to this project. A 3-year 'Proof of Concept' project focused on two activities: controlling rats and feral cats by intensive trapping within a 100 ha trial area at Thoven, and reducing feral pig impacts on gardens on the flanks of Mont Panié. The rat and cat trapping trial was initiated to determine the feasibility of trained local trappers reducing these pests to low densities. Trapping also allowed for rodent species to be identified and their densities to be assessed. Experienced pest control staff from the DOC provided advice and support in the field, trained local trappers and assisted in data analysis and evaluation.

As a result of the commitment and skills of the local trappers rat control targets were met, leading to some confidence that rat and cat control is achievable on Mont Panié-proper. A larger-scale experimental control project focusing initially on rats and cats is currently being designed to be undertaken within the reserve. If successful, it can be anticipated that this experiment will lead to extensions to the control regime on Mont Panié as well as the initiation of further intensive small mammal control projects elsewhere in New Caledonia. While there are important differences between New Caledonia and NZ the successful control of invasive mammalian predators in Mont Panié forests will result in important biodiversity conservation outcomes, as has occurred in NZ. Community capacity to trap rats may also lead to their reduced impacts on corn and some other garden crops.

In addition to pest control activities to achieve biodiversity conservation outcomes, there is strong support for the development of local capacity to more effectively reduce feral pig impacts in tribal gardens. Many local people are dependent on garden produce for their food and income. Depredations by feral pigs in gardens have led to significant impacts. The traditional practice of growing water taro in terraces has been abandoned. Pigs are now seriously threatening yam cultivation. Yams are a staple in the diet of local people and a pillar of the Kanak culture. Other crops are also being affected. Attempts to reduce pig impacts have included trapping and fencing-off gardens, and burning vegetation surrounding gardens in the hope that pigs will not venture into these open areas. None of these initiatives have been particularly successful. At the start of the 'Proof of Concept' project the President of Province Nord, M. P. Neaoutyine, encouraged the project team to find a solution to the pig problem, suggesting that reducing pig impacts in gardens would lead to important social and economic benefits.

Following a visit by experienced pig hunters from NZ, it was resolved that use of experienced

local hunters and trained dogs was likely to be a key tool, perhaps alongside others, in reducing pig impacts in gardens in the Mont Panié area. Three trained and experienced pig-hunting dogs were brought to Mont Panié and, with their NZ handlers, quickly demonstrated that this technique, indeed, has potential to reduce the pigs-in-gardens problem. The three-dog team (reduced to 2 dogs midway through the 2-month visit when one was killed by feral dogs while holding on to a pig) caught just over 1 pig per hunting day. This capture rate is much higher than what local hunters had been able to achieve using local dogs. A growing number of local men accompanied the NZ hunters on hunting trips to learn first-hand how dogs can be used to effectively catch pigs. By the end of the NZ pig hunters' visit there was strong interest from some local hunters to acquire, train, and hunt with dogs using the techniques demonstrated. Interest was also expressed, and some calls received, for such hunter-dog teams to respond to requests to remove pigs from specific gardens.

This investigation showed that experienced local hunters using trained dogs could potentially address the problem of pig impacts in tribal gardens. While there are still significant challenges to overcome, including identifying and addressing the mechanism which leads to widespread blindness in dogs in the area and the risk of feral dogs disrupting hunting activities, there is very strong local support to develop this approach further. It is anticipated that addressing social and economic objectives by controlling feral pigs around tribal gardens will also lead to further support for biodiversity conservation activities further up the mountain within the reserve. A range of other benefits for people, dogs and native wildlife can also be expected.

With positive results from both the rat and cat trapping at Thoven and from the pig hunting using trained dogs around gardens, discussions are now focused on designing projects to advance biodiversity conservation objectives using intensive experimental predator control at selected sites within the Mont Panié reserve, and establishing a garden protection service to be provided by local hunters with trained dogs using proven techniques. Further meetings have been held with stakeholder groups to report on progress, and with government agencies and other organizations with interests and capacity to support the next phase in this capacity-building programme.

## CAT, DOG AND RAT ERADICATION, VIWA ISLAND, FIJI

A proposal by researchers from the University of the South Pacific (USP) to eradicate invasive cane toads (*Bufo marinus*) from Viwa Island in Tailevu, Fiji, to protect the endangered Fijian ground frog (*Platymantis vitiana*) was modified following consultations with local residents to first include the eradication of cats, dogs and rats. The proposal to eradicate cane toads and invasive mammals (rats, feral cats, and feral dogs) was in line with objectives in the Fiji Biodiversity Strategy and Action Plan (1999) to protect and conserve Fiji's biodiversity. Managing the threats posed by invasive organisms is considered a high priority because of the significant effects these pest species have on Fiji's fragile insular ecosystems. Both rats and cane toads are listed as problem species in the strategy.

Eradicating invasive species is a challenging task, especially on inhabited islands. With no record of cane toads having been successfully eradicated anywhere in the world, there was even more reason to advance carefully towards such an outcome on Viwa. It was clear from the outset that in addition to developing new management approaches and techniques, securing the support and engagement of Viwa residents in eradication, and subsequent biosecurity activities, would be critical. Consultation with stakeholders over several years resulted in concerns raised by villagers being addressed in the planning process and implemented as part of the eradication programme. These included water supply, health issues, rubbish disposal, and crop losses to rats.

The essence of the Viwa project is that local residents would undertake cat, dog and rat eradication and on-going biosecurity activities themselves. The development of local capacity to manage invasive mammals would also be a useful step towards the proposed cane toad eradication. Development of this local support and capacity required the community to be involved in all decisions and to be engaged in the full range of management activities. The Pacific Invasives Initiative coordinated a series of meetings and a feasibility study and established a project management team. A project coordinator and two project managers reported to a local stakeholders committee. A Technical Advisory Group (TAG) provided specialist advice from time to time.

The project coordinator is a USP conservation biologist. He was responsible for ensuring that

there was an atmosphere of understanding and trust between all those involved and that everything was in place for a successful outcome. One of the project managers was responsible for facilitating communication between landowners and the heads of the Mataqali (clans), and for managing project staff and their activities on the island. The other project manager was responsible for project logistics, finance, equipment and data collection and analysis. The stakeholder committee included the village chief, the six heads of the Mataqali and a number of women and youths. The TAG consisted of specialists from the PII and the DOC, a local consultant, and government administrators from the Native Lands Trust Board, Fijian Affairs Board and Ministry of Environment.

It is clear that there are important benefits in local residents undertaking an eradication operation. These people have a greater stake in the project, especially where there are anticipated social and economic, as well as biodiversity conservation, outcomes. As a result local residents may be more motivated to achieve project objectives than workers from elsewhere. Another important consideration is that having a skilled, permanent workforce on the island allows effective biosecurity measures to be consistently applied, and for rapid responses to be taken should targeted pests re-invade. An interesting additional benefit from the Viwa rodent eradication project was the promotion by Viwa residents of another rodent eradication operation on a nearby island. There are significant additional challenges in eradicating rats from inhabited islands compared to remote, uninhabited ones. For example, applying toxic baits in and around peoples' homes and managing the risks of rats arriving with freight and luggage present potential problems. Observations from the Viwa project suggest, however, that there may also be important advantages in having resident people present.

Preliminary results from these eradications indicate that feral dogs were successfully removed by local residents without the need for any formal eradication project. Feral cats were informally removed, although we understand one pet cat remains on the island. Once a grid network of cut and marked 'bait lines' had been established over the entire 60 ha island, baits containing the anti-coagulant brodifacoum were repeatedly presented in 1,260 bait stations on this grid until all sign of rat presence declined to zero. Such an effort by a large number of local residents is a major achievement. Observations have already been made of increased numbers of lizards and birds, and people are reporting improved garden yields. It has even been suggested that there are more fish in fishing areas around the island. This is presumably due to people being involved in the rat eradication rather than going fishing! The village development committee is pursuing the construction of a lodge to provide accommodation for eco- and religious tourism. This potential source of income provides some of the impetus for the continuing biosecurity work on the island, but a recent PII survey showed that not having to deal with rats in houses contaminating food, biting children, and eating clothes was recognized as a major outcome of the eradication.

Sign of at least one rat was found during surveillance checks after the eradication was complete. It was not clear whether this indicated the eradication had failed, or that a rat quickly invaded from elsewhere. Contingency measures were immediately put in place, including putting out more bait in the area where the rat sign was found. There has been no further sign of this or other rats.

Some important lessons have been learned from this project about working with, and empowering, local people. These lessons will be important as further demonstration projects are initiated. Local people have skills and knowledge which will be useful to them as they continue to enhance their own environment. Many of these people are now also strong advocates for invasive species management. Print and television media carried stories of the eradication and religious tourists from many countries around the world have seen what can be done when invasive species management projects address both biodiversity and livelihood issues.

## RESTORING VATU I RA ISLAND, FIJI

Vatu I Ra is a small (2.3 ha) island located about 15 km off the north-eastern coast of the main island of Viti Levu. The island is uninhabited but is visited by fishermen who are known to harvest seabirds and eggs. The island has also been used for periodic tourist visits, from which the land-owning community has not derived benefits. Vatu I Ra has been identified by BirdLife International Pacific (BIP) as an Important Bird Area as it supports a large breeding colony of black noddies (*Anous minutus*). Following its declaration as an important bird area, the owners began to work towards establishing an ecotourism business focused on bird

watching on the island. A nearby hotel is assisting in this community venture.

A relationship was initiated in 2003 when the owners of Vat I Ra approached BIP for assistance in managing the island. Two initial surveys and a community awareness meeting were undertaken which led to a proposal by BIP for the rats to be eradicated from the island. There was clear evidence that the rats were preying heavily on seabirds and their eggs and chicks. This proposal was supported by the community. Funding for the rat eradication was provided by the Australian Government's Regional Natural Heritage Program through the Critical Ecosystems Partnership Fund. PII support included assistance to BIP staff in designing the operation and facilitating specialist inputs from an experienced DOC practitioner to refine techniques and to train field staff. Several local people were trained and participated in the eradication operation. The eradication operation involving two hand-spread applications of toxic bait was successful in removing all the rats from the island. In addition to undertaking the eradication itself, local people were also trained in follow-up rat monitoring, quarantine and contingency measures, and have participated in subsequent bird surveys. These people have also learned much about seabirds and other animals and plants on the island.

A lot of effort has been applied to promote Vatu I Ra as an important bird island globally and nationally. Within the community BIP held meetings to raise awareness and generate support. A sign board was erected on the island to discourage fisherman from harvesting seabirds and their eggs. It has been very encouraging to see that local landowners have visited the nearby school to talk about Vatu I Ra, its biodiversity, and the need to protect it. They have also given presentations in district and provincial meetings to inform the whole province about the importance of keeping the island rat-free.

One important community impact has been that it has brought together the members of Mataaqali (clans) from several villages (especially Nasau and Navuniivi villages) as well as people from urban areas. According to the clan leader, this to him is the greatest thing that could ever happen in his lifetime; his people have realized what is happening in the community and have all come together with their families to Vavia (site of eco-lodge) to assist in seeing that the project is successful. As a side benefit of the project, after realizing the number of youth they have in the clan, they have set up a Development Youth Group.

Two government ministries (Tourism and Youth) have recognised the initiative of the landowners and have assisted them financially, as well as in providing materials. The Ministry of Youth donated farming equipment and musical instruments to encourage the youth to farm their land and to record local songs as a source of alternative income.

The development of new insights and skills, the engagement of new networks and the establishment of initiatives following the rat eradication are all seen as steps towards the community managing the island in a sustainable manner.

Following the success of the Vatu I Ra eradication project BIP has received a number of requests for biodiversity surveys and rat eradication from other provinces in Fiji. Funds have recently been secured from the David and Lucile Packard Foundation to identify Fiji's important seabird areas and to study the feasibility of undertaking further eradications.

As these and other projects evolve, it is becoming increasingly clear that identifying social and economic objectives, as well as biodiversity conservation ones, from the project design stage is important. Exciting opportunities to more effectively manage invasive species to reduce their impacts on native biodiversity, and on peoples' livelihoods, can be anticipated if integrated objectives are tackled in partnership.

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# GLOBAL HARMONISATION IN THE FIELD OF INVASIVE SPECIES MANAGEMENT PRODUCT DEVELOPMENT

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***Abstract:*** Problems associated with managing vertebrate invasive species, defined as any exotic or native species that has spread beyond its natural range, are remarkably common around the world. Although the species may differ, the niches they fill and the damage they generally cause is often consistent. As such, a possibility exists for greater collaboration and harmonisation in developing new tools to manage the impacts of invasive species. Moreover, the considerable expense of developing new products, the often onerous registration process, and the lack of return on investment leading to market failure has meant that progress within the field of invasive animal management product development can sometimes be stifled. This paper details a potential way forward using specific examples of ways the Invasive Animals Cooperative Research Centre (IACRC), Pestat P/L, and other IACRC commercial partners aim to provide overseas organisations with humane non-lethal and lethal wildlife management tools. Conversely, we detail overseas-developed products currently being tested in Australia, and a proposal to keep abreast of new developments in other countries to ensure invasive species management within Australasia remains of the highest level. This can only be achieved through truly collaborative research and the global harmonisation of registration packages, so product development costs can be minimised and the necessary scales of production can be attained.

***Key Words:*** collaboration, harmonisation, Invasive Animals CRC, invasive species, pesticides, product development, registration.

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## INTRODUCTION

The World Conservation Union (IUCN) defines invasive species as *organisms (usually transported by humans) which successfully establish themselves in, and then overcome, otherwise intact, pre-existing native ecosystems* (www.issg.org). This paper, however, takes a broader definition, and one that is consistent with that of the Australasian Invasive Animals Cooperative Research Centre (IACRC). In the context of the IACRC, "invasives" refers to terrestrial and freshwater vertebrates, including overabundant native species. While not always transported by humans (although many have been accidentally or deliberately introduced to areas beyond their natural range), overabundant native species have generally become such through a direct result of habitat changes that positively influence that species' ability to survive and reproduce. The IACRC aims to counteract the impact of invasive animals through the development and application of new technologies and by integrating approaches across agencies and jurisdictions (Saunders et al. In Press).

Currently, Australia is host to 56 invasive vertebrate animal species. Most agricultural sectors suffer significant economic losses through factors such as predation of livestock, crop damage, and competition for feed by invasive species. Furthermore, in the last two centuries, 27 indigenous mammal species have become extinct in Australia, accounting for over half the mammalian extinctions in the world over that time period. The invasive red fox (*Vuples vulpes*), feral cat (*Felis catus*) and European rabbit (*Oryctolagus cuniculus*) are three species that have significantly contributed to this unenviable record.

It has been estimated that invasive animals, in particular European rabbits, wild dogs (*Canis lupus familiaris*), red foxes, feral pigs (*Sus scrofa*), and feral cats, cost Australia at least AUS$720 million annually through environmental, economic, and social damage (McLeod 2004). Furthermore, controlling feral animals costs governments and landholders more than AUS$60 million a year, with an additional AUS$20 million spent annually on research to find better methods of management.

Whether in the United States (US), Australia or New Zealand (NZ), all countries that are heavily impacted by invasive species, development of invasive animal management products is generally undertaken for one of four reasons: (1) managing an invasive species population through fertility or lethal means; (2) reducing the impact of introduced predators on livestock and endangered species; (3) preventing damage to infrastructure, fields and crops by herbivores, birds, and burrowing pests; (4) reducing the risk of disease transmission among invasive species, domestic animals, and humans, or (5) detecting and preventing the establishment d/or spread of new invasive species. Such products are commonly based on managing the invasive species populations through fertility control, lethal means, or exclusion.

Most invasive mammals occur in multiple countries and, therefore, invasive animal management products should have multiple or global markets. Table 1 details the continents and countries that identify the mammals listed as invasive in the Global Invasive Species Programme's *100 of the World's Worst Invasive Alien Species*. As can be seen, the brushtail possum (*Trichosurus vulpecula*) in NZ is a rare exception to the rule that most invasive species occur in multiple locations throughout the world.

## HARMONISATION

Harmonisation is defined by Wikipedia as the process in international law by which different states adopt the same laws (en.wikipedia.org/wiki/Harmonisation). Many forms of global harmonisation are essential and well underway. Relevant examples include the International Cooperation on Harmonisation of Technical Requirements for Registration of Veterinary Medicinal Products (VICH) and the United Nations Economic Commission Globally Harmonized System of Classification and Labeling of Chemicals. In relation to the second, Governments endorsed Chapter 19 of Agenda 21 at the United Nations Conference on Environment and Development in 1992 as the international program of action for developing and implementing national chemical safety and management programs.

**Table 1.** Global distribution of the mammal species listed within the *100 of the World's Worst Invasive Alien Species list*. Data collated from the Global Invasive Species Database (www.issg.org/database/welcome).

| Common name | Scientific name | Continents and countries in which they are considered invasive |
|---|---|---|
| Brushtail possum | *Trichosurus vulpecula* | New Zealand |
| European rabbit | *Oryctolagus cuniculus* | Australia, Canada, Chile, France, Kiribati, Mauritius, Mexico, New Zealand, North America |
| Feral cat | *Felis catus* | All continents, except Antarctica, and many islands |
| Feral goat | *Capra hircus* | Americas, Australia, Europe |
| Feral pig | *Sus scrofa* | All continents, besides Antarctica |
| Grey squirrel | *Sciurus carolinensis* | Canada, Ireland, Italy, South Africa, United Kingdom |
| House mouse | *Mus musculus* | All continents, except Antarctica, and many islands |
| Indian mongoose | *Herpestes javanicus* | Americas, Asia, South Pacific islands |
| Long-tailed macaque | *Macaca fascicularis* | Hong Kong, Indonesia, Mauritius, Palau |
| Nutria | *Myocastor coypus* | North America, Europe, Africa |
| Red deer | *Cervus elaphus* | Australia, Israel, New Zealand, South America, Virgin Islands |
| Red fox | *Vulpes vulpes* | Australia, Canada, Mexico, United Kingdom, North America |
| Ship rat | *Rattus rattus* | All continents, except Antarctica, and many islands |
| Stoat | *Mustela erminea* | North America, Europe, Asia, New Zealand |

Chapter 19 program areas include: (1) expanding and accelerating international assessment of chemicals risk, (2) harmonisation of classification and labeling of chemicals, and (3) information exchange on toxic chemicals and chemicals risks, among other areas (www.nicnas.gov.au/International/Agenda_21.asp). As such, harmonisation is already occurring in the international classification of toxins. Of particular interest to the authors is the harmonisation of data requirements for registering toxins or other actives for the purpose of invasive species management.

In relation to invasive species management products the potential benefits of global harmonisation are: (1) standardized regulatory information processing worldwide, (2) reduced costs of product development, and, in turn, of products, (3) potential for regulatory burden sharing, (4) completion of studies that may not be appropriate to or affordable in other countries, (5) accelerated time-to-market, (6) reduced duplication and additional use of animals/resources, and (7) achieve economies of scale that make product development and markets viable.
The authors do, however, recognize that although generally a positive step, international harmonisation does need to be balanced with national needs and concerns.

The cost of developing and registering new invasive animal products is only envisaged to become more expensive and complex. Examples of product development and registration costs in Australia range from AUS$500,000 and 4 years for the relatively simple PIGOUT® feral pig bait to potentially AUS$5M and 5+ years for the more complex registration (in preparation) of para-aminopropriophenone, a predacide currently under development at the IACRC (see below). Additionally, good scientific research is no guarantee of success, as shown with the research into virally-vectored immuno-contraception for rabbits, foxes and house mice (*Mus domesticus*), which cost more than AUS$10M over 12 years, but resulted in no registered products (Williams 1997). A similar large-scale genetic fertility modification project (known as daughterless technology) is currently underway with European carp (*Cyprinus carpio*) in Australia (Thresher and Bax 2003, Grewe et al. 2005), with the project estimated to cost more than AUS$10M over 10 years, albeit with the prospect of success still achievable. In addition, vertebrate pesticide registrations are actively being lost due to the cost and data requirements in maintaining them (Jacobs 1992).

Registering vertebrate pesticides already has commonalities worldwide. Whether through the Australian Pesticide and Veterinary Medicine Authority (APVMA), the US Environmental Protection Agency (EPA), or the NZ Environmental Risk Management Authority (NZ ERMA), data requirements for registering a new active or re-registering an existing active are somewhat analogous. Table 2 documents the respective registration requirement outlines for the APVMA and EPA. Although similar, data requirements by the EPA are possibly more extensive, and this potentially is a serious drawback to direct adoption of the EPA model in the event of product registration harmonisation.

## PRODUCT DEVELOPMENT

The initial IACRC product pipeline is detailed in Table 3 and it indicates the most promising products currently under development at the IACRC. The list is not exhaustive, and numerous other products for canids, feral pigs, cane toads (*Bufo marinus*) and European carp are in various stages of development. Below are some further details on each product.

### Feral Pig Bait, New Toxin and Delivery Systems

Development of PIGOUT®, a non-meat-based omnivore bait designed to be attractive to feral pigs but not to herbivores or carnivores, began in 2004 through the support of Meat and Livestock Australia Ltd. and the National Feral Animal Control Program (NFACP). The manufactured bait had to be highly attractive to pigs, cheap, target-specific and easy to use. Trials have been conducted around Australia, and have achieved 78 ± 4% (S.E., n=9 sites) population or activity reduction with ground baiting and 62 ± 9% (S.E., n=4 sites) biomarked or activity reduction with aerial baiting (see Cowled et al. 2006a for trial example). High target-specificity has been achieved in all Australian trials (Cowled et al. 2006b). A registration package was submitted to the APVMA in August 2006 and the product is expected to be registered shortly. PIGOUT® has also been investigated for its ability to deliver simulated vaccines both in Australia (Cowled et al. In Press) and in the US (Campbell et al. 2006, Campbell and Long 2007). Further trials have been conducted in NZ and the United Kingdom (UK).

Toxic PIGOUT® baits currently contain sodium monofluoroacetate (1080). Although 1080 is lethal to feral pigs, large doses are required and its

**Table 2.** Registration dossier requirements for a new active by the Australian Pesticide and Veterinary Medicine Authority (from www.apvma.gov.au/industry/MORAG.shtml) and the EPA (from www.epa.gov/pesticides/regulating/data.htm).

| APVMA New Active Registration Dossiers | EPA Pesticide Data Requirements |
|---|---|
| Part 1 Application overview | **Types of Studies:** |
| | Residue chemistry |
| Part 2 Chemistry and manufacture | Environmental fate |
| | Degradation studies |
| Part 3 Toxicology | Metabolism studies |
| | Mobility studies |
| Part 4 Metabolism and kinetics | Dissipation studies |
| | Accumulation studies |
| Part 5A Residues | |
| | **Hazard to Humans and Domestic Animals:** |
| Part 5B Overseas trade aspects of residues in food commodities | Acute studies |
| | Subchronic studies |
| | Chronic studies |
| Part 6 Occupational health and safety | Teratogenicity and reproduction studies |
| | Mutagenicity studies |
| Part 7 Environment | Metabolism studies |
| | Reentry protection |
| Part 8 Efficacy and safety | Pesticide spray drift evaluation |
| Part 9 Other trade aspects | **Hazard to Nontarget Organisms:** |
| | Short-term studies |
| Part 10 Special data | Long-term and field studies |
| | Product performance |

**Table 3.** The Invasive Animals Cooperative Research Centre (IACRC) product pipeline by year. The table lists some of the key products currently being developed with the IACRC, all of which are potentially suitable for global markets.

| Product | 2006 | 2007 | 2008 | 2009 | 2010 | 2011 | 2012 |
|---|---|---|---|---|---|---|---|
| PIGOUT- feral pig bait* | | | | | | | |
| Freeze-dried calici-virus bait for rabbits | | | | | | | |
| High output carbon monoxide fumigator for burrowing animals* | | | | | | | |
| PAPP fox bait | | | | | | | |
| PAPP wild dog bait* | | | | | | | |
| Attractants/tools for feral pig management | | | | | | | |
| A new humane feral pig toxin | | | | | | | |
| Daughterless carp and other carp control tools | | | | | | | |

* Already trialled overseas

effect can be variable. This has lead to a search for new feral pig actives. A promising candidate has recently been found that has been shown to be a rapidly lethal and humane means of euthanizing feral pigs in gavage and bait delivery pen trials. The compound will be the subject of research, development and registration efforts over the next three years. Overseas markets, such as NZ and the US (Hawaii in particular), will be sought for HOG-GONE®, the new active contained within the PIGOUT® bait, so beneficial scales of production can be achieved.

**Freeze-Dried Calici-Virus Bait**

Wild European rabbits in Australia exhibited the fastest rate of spread of any colonizing mammal anywhere in the world. They now represent one of the most widely distributed and abundant mammals in Australia. Rabbit control has been greatly assisted by the release of two viruses as biological control agents, myxoma virus and rabbit hemorrhagic disease virus (RHDV) or Calici-virus. Compared to myxoma virus, RHDV is a very efficient and humane method of rabbit control, and it has had a significant impact on some (but not all) rabbit populations, e.g., a mean 92% population reduction in Victoria since its release (McPhee et al. 2001). Until recently, the virus was spread following the injection of a small number of captured and re-released rabbits. However, approval has now been obtained from APVMA for the spread of RHDV by the oral route on treated carrots or grain. Despite being a step in the right direction, there have been unforeseen difficulties in dealing with the frozen viral suspension that directly impact on the efficient and effective deployment of RHDV in the field and, therefore, the efficacy of this bio-control. The greatest impediment to the widespread distribution of the stock RHDV virus is the need to store and distribute the finished product at ultra-low temperatures. As a result of changes to transport regulations, shipments containing dry ice are classified as "Dangerous Goods" and their movement, especially by airfreight, is exceedingly difficult. The result is increased cost, in addition to failure of some shipments.

The current project, supported by the NFACP, aims to remove these impediments and difficulties in producing, storing, shipping and handling the viral suspension by producing a stock of virus in a freeze-dried form. Therefore, this project has three complementary aims. Firstly, to dramatically improve the ease and convenience of manufacturing and storing a shelf-stable RHDV product that will reduce the costs of goods. Secondly, to increase the shelf-life and simplify the storage and shipping conditions of the RHDV product that will reduce the cost of goods to the end-user. Thirdly, to produce a highly practical, shelf-stable RHDV product that will increase market uptake and responsible use within an integrated rabbit control strategy that will enhance the overall effectiveness of RHDV as a bio-control agent and prolong its effective life. The end product will be suitable for export to other countries or islands wanting to manage or eradicate European rabbits.

**High Output Carbon Monoxide Fumigator**

The control of rabbits has historically been achieved by a number of methods, with warren fumigation being one method that is considered to be a simple and effective technique, as well as one that can be readily undertaken by landholders. Pressure fumigation is regarded as being more efficient than the static method, due to its greater ability to force gas throughout the warren with all openings found and sealed. However, in recent times, there have been Occupational Health and Safety issues concerning the use of the currently used fumigant (chloropicrin) with pressure fumigators that have halted the use of this technique by landholders in New South Wales, Australia. Also, animal welfare concerns have arisen over the use of chloropicrin in general. Carbon monoxide (CO) has been proposed as the most humane gas to use for fumigation purposes, though to date there has been no successful means developed to deliver sufficient concentrations and purity of this gas into a warren system. The Victorian Department of Primary Industries, through NFACP support, has been developing a highly portable fumigator which can be carried by a single person (Gigliotti et al. 2001). Carbon monoxide fumigation is currently registered in Australia, NZ, the US, and the UK. The prototype high output, fan-forced CO pressure fumigator is particularly suitable for large and complex burrows. Rabbits in Australia and NZ, pocket gophers in the US (Fagerstone 1997) and European badgers (*Meles meles*) in the UK all occupy dwellings that potentially require pressure fumigation if CO levels are to increase rapidly enough to cause the quick and humane death of burrow occupants.

**Para-aminopropriophenone**

Para-aminopropriophenone (PAPP) was initially investigated as an alternative predacide to 1080 by the US Department of Agriculture (USDA) 25 years ago (Savarie et al. 1983). Research into PAPP was discontinued following re-approval of 1080 use in Livestock Protection Collars. The mode of action for PAPP is the conversion of haemoglobin to methaemoglobin, the latter of which cannot carry oxygen. A rapid and humane death results from a lack of oxygen to the brain and cardiac muscles. The average time to death for bait-delivered PAPP is approximately one hour for feral cats and foxes and 2 hours for wild dogs (D. Dall, personal observation). Low levels of methaemoglobin reductase in eutherian carnivores may be a factor that is responsible for their high sensitivity (compared to other mammals) to the compound (Srivastava et al. 2002). PAPP is currently being investigated for wild dogs and foxes (Fleming et al. 2006, Lapidge et al. 2006) and feral cats (Fisher et al. 2001) in Australia, and stoats (*Mustela erminea*), ferrets (*M. putorius*), and feral cats in NZ (Fisher et al. 2005, Murphy et al. 2005, Fisher and O'Conner 2007). The canid component of the project is sponsored by Australian Wool Innovation Ltd. (AWI), and it is hoped that a fox PAPP bait registration will be submitted late in 2007 with a wild dog PAPP registration to follow thereafter. The active is suitable for coyotes (*Canis latrans*) in the US and other pest or invasive eutherian carnivores requiring humane lethal control.

**'Daughterless' Technology**

European carp are often referred to as the 'rats of the river' in Australia. The species is widespread throughout the Murray-Darling Basin, Australia's most extensive and important water source, and causes extensive environmental problems in relation to water quality and native fish survival. Genetic technologies offer potential to manage carp populations. This has lead to the study of sex-specific apoptosis in carp and other model species for achieving 'daughterless' fish (Grewe et al. 2005). The Commonwealth Scientific and Industrial Research Organization (CSIRO) commenced research into daughterless carp in 2003 with the backing of the Murray-Darling Basin Commission and the Pest Animal Control CRC (IACRC's forerunner). Daughterless technology involves an engineered genetic construct using homologous species-derived genes that are inheritable and that bias offspring sex ratios towards males. Models indicate that replacing 5% of wild type recruits each year with daughterless carriers would lead to a significant decrease in population levels by 2020 and near extinction by 2030 (Thresher and Bax 2003). The technique is also being investigated for cane toads in Australia, and is potentially useful for other invasive fish or amphibians worldwide.

**Attractants and Pheromones**

Numerous attractants, pheromones, and repellents are currently being investigated within the IACRC, Pestat P/L, and research partners for commercial or environmental applications. With AWI's assistance, the first product to be commercialized by Pestat P/L has been FeralMone®, a highly attractive dog and fox lure that is based on a proprietary formulation of synthetic fermented egg (Bullard et al. 1978). Field trials in Australia showed that FeralMone® significantly increased site (bait or trap) visitation and bait take (Hunt et al. In Press). Other lures are currently being investigated for feral pigs and carp (attractants), cane toads (attractants and repellants) and kangaroos (*Macropus* spp., repellents). It is likely that each would be useful in multiple countries around the world for various wildlife management applications. Ideally, different countries and markets need to be identified early in this process so appropriate packaging and labeling can be developed.

**USDA Products Currently Being Investigated in Australia**

While Australia is one of the regions at the forefront of invasive species management product development (perhaps out of necessity), we also constantly look to our international research partners in NZ (Connovation Ltd., NZ Department of Conservation, and Landcare Research), the US (USDA National Wildlife Research Centre and University of Minnesota) and the UK (Central Science Laboratory and University of York) for further advances in the field. Invasive species products previously developed by our partners (or other organisations) that are currently under investigation in Australia, besides those previously mentioned, include the GnRH GonaCon™ immunocontraceptive vaccine (Fagerstone et al. 2006) and the SenesTech Inc. accelerated ovarian senescence product for reducing the fertility of macropods and other invasive species. This research is being undertaken within the IACRC. Pestat P/L, as a separate entity, is also investigating

DRC-1339 (aka Starlicide™) for European starling (*Sturnus vulgaris*) control (Lapidge et al. 2005); M-44 mechanical ejectors for red fox and wild dog control (Marks et al. 2003, 2004); the Coyote Operative Lure Device (CLOD) for wild dog and fox control (Berentsen et al. 2006); and egg oiling for bird control (Martin et al. 2007). Furthermore, the IACRC is interested in the cocoa- and coffee-derived methylxanthines for canid control in the event that a further predacide is required in addition to PAPP (Fagerstone et al. 2004, Johnston 2005). All of the products will require preparation of extensive registration packages before they can be used in Australia. For the most part, such registration packages exist or are in preparation overseas, and it is hoped that future harmonisation between the relevant national pesticide registration organizations will facilitate product registration by the APVMA in Australia.

## DISCUSSION

Product development in the field of invasive species management is currently evolving at a rapid pace never seen before. As governments and industry begin to appreciate the scale of the ever-increasing invasive animal problem, a growing number of biotechnology-based start-up companies are jostling to establish positions in what are, to a degree at least, niche markets. Despite this activity, it is likely that the role of product development will heavily rest with governments or semi-government organizations working with industry partners (the basis of the Cooperative Research Centres in Australia). The high cost of product development and registration, the small scale on which most vertebrate pesticides are used, together with the diminutive associated profit margins are likely to demand this for the foreseeable future. We suggest that the most effective way of reducing product development costs, times to availability, and regulatory burdens is through global harmonisation of invasive species management products. As outlined in this paper, we are now heading in the right direction. International efforts are currently being coordinated on wildlife fertility control research. Development of the new predacide PAPP is an ongoing collaboration between Australia, NZ and the US. PIGOUT® feral pig baits have undergone extensive testing in Australia, the US, the UK, and NZ, and the CO pressure fumigator has also been tested in multiple countries. The question we pose here is, "Are we doing all we can to facilitate product registration harmonisation where appropriate and possible?" Self-evidently, this must involve obtaining data for registrations that meet international regulatory standards, and also ensuring that any unique information required by specific international regulators is obtained or at least highlighted.

## CONCLUSION

Invasive species are a global problem and global solutions are required. Effective international collaboration is essential, both between researchers and government regulatory agencies alike. This, of course, must be done with appropriate sensitivity to intellectual property ownership and appropriate catering for commercial participants to ensure private investment in the field of invasive species management continues to strengthen. Ideally, a formal agreement is required between international regulatory agencies, such as the APVMA, US EPA, and NZ ERMA, before this process can occur, as has transpired between governments in relation to chemical classification and labeling. We would also suggest that a Global Invasive Species Management Product Research Register would be particularly constructive, so researchers, product developers, private companies and governments can readily assess efforts that have already been made towards developing and registering particular products so as to avoid unnecessary duplication of effort, cost, and use of animals for research purposes.

## LITERATURE CITED


BERENTSEN, A. R., J. J. JOHNSTON, R. E. MAULDIN, AND R. H. SCHMIDT. 2006. Using the CLOD to deliver pentachlorobenzene to coyotes (*Canis latrans*). Proceedings of the Vertebrate Pest Conference 22:277-281.

BULLARD, R. W., S. A. SHUMAKE, D. A. CAMPBELL AND F. J. TURKOWSKI. 1978. Preparation and evaluation of a synthetic fermented egg coyote attractant and deer repellent. Journal of Agricultural Food Chemistry 26:160-163.

CAMPBELL, T. A., S. J. LAPIDGE AND D. B. LONG. 2006. Baits to deliver pharmaceuticals to feral swine in southern Texas. Wildlife Society Bulletin 34:1184-1189.

CAMPBELL, T. A. AND D. B. LONG. 2007. Species-specific visitation and removal of baits for delivery of pharmaceuticals to feral swine. Journal of Wildlife Diseases 43:485-491.

COWLED, B. D, E. GIFFORD, M. SMITH, L. STAPLES AND S. J. LAPIDGE. 2006a. Efficacy of manufactured PIGOUT® baits for localized control of feral pigs in the semi-arid rangelands in western Queensland. Wildlife Research 33:427-437.

COWLED, B. D., S. J. LAPIDGE, M. SMITH AND L. STAPLES. 2006b. Attractiveness of a novel omnivore bait, PIGOUT®, to feral pigs (*Sus scrofa*) and assessment of risks of bait uptake by non-target species. Wildlife Research 33:651-660.

COWLED B. D., S. J. LAPIDGE, M. SMITH AND L. STAPLES. In press. Vaccination of feral pigs (*Sus scrofa*) using iophenoxic acid as a simulated vaccine. Australian Veterinary Journal.

FAGERSTONE, K. A. 1997. Overview of controls: why they work and how they function: toxicants. Pages 17-24 *in* D. L. Nolte and K. K. Wagner, Wildlife Damage Management for Natural Resource Managers.

FAGERSTONE, K. A., J. J. JOHNSTON AND P. J. SAVARIE. 2004. Predacides for canid predation management. Sheep & Goat Research Journal 19:76-79.

FAGERSTONE, K. A., L. A. MILLER, K. S. BYNUM, J. D. EISEMANN AND C. YODER. 2006. When, where and for what wildlife species will contraception be a useful management approach? Proceedings of the Vertebrate Pest Conference 22:45-54.

FISHER, P., C. A. MARKS AND M. JOHNSTON. 2001. New techniques for feral cat management: biological, chemical or fertility control? Proceedings of the Australasian Vertebrate Pest Conference 12:150-155.

FISHER P. M., C. E. O'CONNOR AND E. C. MURPHY. 2005. Acute oral toxicity of p-aminopropiophenone to stoats (*Mustela erminea*). New Zealand Journal of Zoology 32:163-169.

FISHER P. AND C. O'CONNOR. 2007. Oral toxicity of p-aminopropiophenone to ferrets. Wildlife Research 34:19-24.

FLEMING, P. J. S., L. R. ALLEN, S. J. LAPIDGE, A. ROBLEY, G. R. SAUNDERS AND P. C. THOMSON. 2006. A strategic approach to mitigating the impacts of wild canids: proposed activities of the Invasive Animals Cooperative Research Centre. Australian Journal of Experimental Agriculture 46:753-762.

GIGLIOTTI, F., C. A. MARKS AND F. BUSANA. 2001. Development of a carbon monoxide fumigation technique for the control of European rabbits (*Oryctolagus cuniculus*). Proceedings of the Australasian Vertebrate Pest Conference 12:215.

GREWE, P., N. BOTWRIGHT, J. BEYER, J. PATIL AND R. THRESHER. 2005. Sex-specific apoptosis for achieving daughterless fish. Proceedings of the Australasian Vertebrate Pest Conference 13:59.

HUNT, R., D. DALL AND S. J. LAPIDGE. In press. Effect of a synthetic lure on site visitation and bait uptake by foxes (*Vulpes vulpes*) and wild dogs (*Canis lupis dingo/Canis lupis familiaris*). Wildlife Research.

JACOBS, W. W. 1992. Vertebrate pesticides no longer registered and factors contributing to loss of registration. Proceedings of the Vertebrate Pest Conference 15:142-148.

JOHNSTON, J. J. 2005. Evaluation of cocoa- and coffee-derived methylxanthines as toxicants for the control of pest coyotes. Journal of Agricultural and Food Chemistry 53:4069-4075.

LAPIDGE, S., D. DALL, J. DAWES, J. TRACEY, R. SINCLAIR AND A. WOOLNOUGH. 2005. STARLICIDE®: The benefits, risks and industry need for DCR-1339 in Australia. Proceedings of the Australasian Vertebrate Pest Conference 13:235-238.

LAPIDGE, S., D. DALL, R. HUNT, B. COWLED, M. SMITH AND L. STAPLES. 2006. A review of the impact of sheep predators in Australia and new control methods under development. Proceedings of the Vertebrate Pest Conference 22:258-263.

MCLEOD, R. 2004. Counting the cost: impact of invasive animals in Australia, 2004. Cooperative Research Centre for Pest Animal Control, Canberra, Australia.

MCPHEE, S., T. BLOOMFIELD, P. SANDELL AND C. MARKS. 2001. Monitoring the impact of rabbit haemorrhagic disease virus (RHDV) in the State of Victoria. Proceedings of the Australasian Vertebrate Pest Conference 12:51-55.

MARKS, C. A., F. GIGLIOTTI AND F. BUSANA. 2003. Field performance of the M-44 ejector for red fox (*Vulpes vulpes*) control. Wildlife Research 30:601–609.

MARKS, C. A., F. GIGLIOTTI AND F. BUSANA AND M. JOHNSTON. 2004. Fox control using a para-aminopropiophenone formulation with the M-44 ejector. Animal Welfare 13:401-407.

MARTIN, J. M., K. FRENCH AND R. E. MAJOR. 2007. The pest status of Australian white ibis (*Threskiornis molucca*) in urban situations and the effectiveness of egg-oil in reproductive control. Wildlife Research 34:319-324.

MURPHY, E., A. LAVRENT, D. MACMORRAN, L. ROBBINS AND P. ROSS. 2005. Development of a humane toxin for the control of introduced mammalian predators in New Zealand. Proceedings of the Australasian Vertebrate Pest Conference 13:137-142.

SAUNDERS, G., S. LAPIDGE, W. FULTON, E. MURPHY, S. SARRE, C. BULLER AND T. PEACOCK. In press. The Invasive Animals CRC: a new research initiative for managing some old problems. Australian Zoologist.

SAVARIE, P. J., P. H. PAN, D. J. HAYES, J. D. ROBERTS, G. L. DASCH, R. FELTON AND E. W. JR SCHAFER. 1983. Comparative acute oral toxicity of para-aminopropiophenone. Bulletin of Environmental Contamination and Toxicity 30:122-126.

SRIVASTAVA S., A. S. ALHOMIDA, N. J. SIDDIQI, S. K. PURI AND V. C. PANDEY. 2002. Methemoglobin reductase activity and in vitro sensitivity towards oxidant induced methemoglobinemia in Swiss mice and Beagle dogs erythrocytes. Molecular and Cellular Biochemistry 232:81-85.

THRESHER, R. AND N. BAX. 2003. The science of producing daughterless technology; possibilities for population control using daughterless technology; maximizing the impact of carp control. Pages 19-24 *in* K. L. Lapidge, editor. Proceedings of the National Carp Control Workshop, March 2003, Canberra. Cooperative Research Centre for Pest Animal Control, Canberra, Australia.

WILLIAMS, C. K. 1997. Development and use of virus-vectored immunocontraception. Reproduction, Fertility and Development 9:169-178.

# MANAGING NON-NATIVE WILDLIFE IN FLORIDA: STATE PERSPECTIVE, POLICY AND PRACTICE

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***Abstract:*** Florida has more non-native wildlife species than any other state: 439 introduced species of fish, wildlife and marine organisms have been observed and at least 123 are established, i.e., highly unlikely to be extirpated without human intervention. Florida is an epicenter for non-native species with a long established pet industry, major tourist attractions, and major ports, primarily Miami. The large number of established species is due to climate match with that of popular tropical pets, habitat disturbance that facilitates invasion, and a depauperate vertebrate fauna in tropical and subtropical portions of Florida. The Florida Fish and Wildlife Conservation Commission's approach to managing non-native wildlife is based on (1) encouraging responsible pet ownership; (2) a regulated industry is preferable to underground traffic; and (3) most introduced species have negligible environmental impacts. Regulations for captive wildlife and non-native aquatic species, first established in the 1970s, employ risk-based bio-security for problematic species, and prohibition of a limited number of species that posed unacceptable risks to the ecosystem, economy, or human health and safety. Effective January 1, 2008, owners of six large reptile species will be required to implant passive integrated transponders to identify individual animals. Although anecdotal evidence suggests dealers have released inventories to establish source populations, the majority of introductions have resulted from release of pets by owners. To close this pathway, a pet surrender network is in the early stages of development. Within the past five years, capacity to detect and manage terrestrial and semi-aquatic species has improved, including surveillance, rapid assessment and response; examples include the Gambian giant pouched rat (*Cricetomys gambianus*), purple swamphen (*Porphyrio porphyrio*), and Argentine black and white tegu (*Tupinambis merianae*).

***Key Words:*** Florida, invasive species, pathways, rapid response, regulations, surveillance.

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## INTRODUCTION

Over 400 species of non-indigenous fish, wildlife and marine organisms have been observed in Florida since its colonization by European settlers in the 16th century. The earliest documented vertebrate introduction dates to 1538 when Hernando de Soto imported European pigs (*Sus scrofa*) as a food source. Apparently, the settlers traded pigs to indigenous humans who allowed these animals to range free, eventually leading to the widespread establishment of one of the more notorious exotic species.

During Florida's long history with non-native species, a $300 million captive wildlife industry has developed, including importers, breeders, and tourist attractions that prominently feature exotic animals. In 2005-06, 3982 facilities were permitted for captive wildlife. Furthermore, Florida is the epicenter of the freshwater tropical fish industry, growing 95 percent of the nation's cultured tropical fish on farms primarily in the Tampa Bay area. Over 800 varieties of tropical fish are cultured in Florida.

In the nascent discipline of invasion biology, there is general acceptance of the so-called 'rule of 10s', which posits that roughly 10% of introduced species become established, and roughly 10% of those become invasive (Williamson 1996, Bomford 2005). Florida's experience is consistent with the general boundaries of this rule. The Florida Fish and Wildlife Conservation Commission (FWC) has adopted risk-based regulations that (1) prohibit the unauthorized release of any non-native species; (2) restrict possession of species that pose substantial environmental or economic risks, or harm to human health; and (3) permit possession of species not deemed to present significant consequences to native fish and wildlife.

With the substantial level of pet ownership, as well as an existing industry based on captive

wildlife and aquarium specimens, FWC's approach to non-native species, reflected in its regulatory framework, is to encourage responsible pet ownership. We believe that a regulated pet industry, allowing environmentally benign species and restricting problematic ones, is preferable to indiscriminate underground traffic that would result from legal bans on the majority of non-native organisms.

## STATUS OF NON-NATIVE SPECIES IN FLORIDA

There are approximately 123 established non-native species in Florida. Established means unlikely to be extirpated without human intervention. For freshwater fishes and marine species, the definition is a subjective assessment based on collective expert opinion. For wildlife, species are considered established if they have reproduced for five or more years. The list includes 22 non-native freshwater fishes (Shafland et al. In Review), 4 amphibians, 36 reptiles, 12 birds, and 18 mammals. There are at least 31 marine species, although this number is an estimate at best, owing to the difficulty in surveying the marine environment and the fact that global ship travel predated virtually all biological surveys. With the exception of the red lionfish (*Pterois volitans*) that is possibly established along Florida's Atlantic Coast, introduced marine species are invertebrates or algae.

Among the freshwater fishes, most introduced species are aquarium releases or escapes from aquaculture, although one species, the pike killifish (*Belanesox belizanus*) escaped from a medical research facility. Of the 22 permanently established species, 13 are in the widely distributed tropical-subtropical Cichlid family and four are South American catfishes. Investigation of introduced fishes in southeast Florida canals has indicated no reductions in absolute abundance or biomass of native species, while the standing crop of all fishes has increased (some due to sterile grass carp [*Ctenopharyngodon idella*] purposefully introduced for vegetation control). Recently, non-native catfishes, sailfin catfish (*Pterygoplythys multiradiatus*) and brown hoplo (*Hoplosternum littorale*) have experienced population expansions over south and central Florida. Study of fish population abundance and diet has not revealed competition or displacement of native fishes. However, sailfin catfish dig spawning burrows, and localized erosion has resulted from this activity. In 2000, a prohibited species, bullseye snakehead (*Channa marulius*) from southeast Asia, were discovered in a canal system in southeast Florida. Six years later, no major changes in fish populations have been observed.

Although only three amphibians are established or potentially established, the Cuban tree frog (*Osteopilus septentrionalis*), introduced in 1931, is now found as far north and west as Tallahassee (with reports in south Georgia). This large, invasive frog is associated with declines attributed to predation in native hylid frogs, including green tree frogs (*Hyla cinerea*) and squirrel tree frogs (*H. squirella*). By contrast, the marine toad (*Bufo marinus*), native to central America and south Texas, garnered much attention following its introduction in 1971, but has had negligible impact on native wildlife and appears to be declining in the southern part of its range in Florida.

Roughly three quarters of the introduced reptiles are now established in Florida. The list of established reptiles is dominated by lizards (31 species), but includes one turtle (red-eared slider [*Trachemys scripta elegans*]), one crocodilian (spectacled caiman [*Caiman crocodiles*]), and three snakes. Of the lizards, most were pet releases from owners or dealers, primarily from the families Iguanidae and Gekkonidae, and have caused minor concern in conservation areas. Several large species are established, including the herbivorous green iguana (*Iguana iguana*) and omnivorous spinytail iguana (*Ctenosaurus similis*) which are considered nuisances by many Floridians, but no evidence exists for ecological damage. The Nile monitor (*Varanus niloticus*), a generalist predator and the largest of the African lizards, is established in a small urban area of southwest Florida, causing concern over potential impacts to a nearby population of burrowing owls (*Athene cunicularia*), a Florida Species of Special Concern. Additionally, Nile monitors are predators of crocodile eggs in Africa, and southward expansion of the Florida population would overlap that of the endangered American crocodile (*Crocodilus acutus*).

In the past decade, considerable attention has been focused on a reproducing population of Burmese pythons (*Python molurus*) in extreme south Florida. The distribution of collected snakes suggests that the population may have been the result of released pets. Other speculation centers on the escape of collections of specimens from dealers during Hurricane Andrew in 1992. Regardless, Burmese pythons span the width of Everglades National Park and are commonly found on adjacent

lands. Although there is no evidence at this time of landscape-level effects, the sociological impact has led to greater restrictions on ownership of this species and several other large reptile species. Less well known is an isolated population of *Boa constrictor* confined to a tropical hammock "habitat island" in Miami. This population has existed since the 1970s with no expansion or other impacts.

Relatively few introduced birds have become established, with 68% having no more than 9 breeding pairs (72 species are not reported breeding), and only 7 species reported from more than 9 of Florida's 67 counties (B. Pranty, personal communication). The largest group of introduced birds are the Psittacines, with over 70 species reported, undoubtedly pet releases, yet only two species are confirmed established. Noteworthy among resident non-native birds is the monk parakeet (*Myiopsitta monachus*), well established in south Florida, which builds large colony nests in electrical transformers causing significant economic impact from power outages and equipment loss. A recent introduction, the purple swamphen (*Porphyrio porphyrio*), has expanded from coastal southeast Florida into the Everglades Conservation Areas, and has been observed on Lake Okeechobee. Its ecological similarity to the native common moorhen (*Gallinula chloropus*) and purple gallinule (*Porphyrula martinica*) have prompted efforts to eliminate this member of the rail family native to central Europe and Asia. One non-native bird species, cattle egret (*Bubulcus ibis*), arrived without human assistance, apparently blown by hurricane winds from nearby Caribbean islands.

Florida's non-native mammals include species with cosmopolitan distributions, such as the house mouse (*Mus musculus*), black rat (*Rattus rattus*) and Norway rat (*R. norvegicus*), as well as four feral species: pig, dog (*Canis familiaris*), cat (*Felis catus*), and goat (*Capra hircus*). Three non-native mammals have become established both by human introduction as well as natural migration: nine-banded armadillo (*Dasypus novemcinctus*), red fox (*Vulpes vulpes*), and coyote (*Canis latrans*), although the latter species' success is at least partially attributed to extirpation of the red wolf (*Canis rufus*) in the early 1900s. Among the more interesting non-native mammals, the rhesus monkey (*Macaca mulatta*) was introduced to a central Florida tourist attraction in the 1930s and now has an established, stable population in river floodplains, as well as a smaller population in south Florida. Similarly, a population of vervet monkeys (*Chlorocebus atheiops*) has been established without apparent consequence in an isolated tropical hammock area within the highly urbanized corridor of Broward County in southeast Florida for 50 years. The Sambar deer (*Cervus unicolor*) has been established on St. Vincent Island in northwest Florida for 100 years; the National Wildlife Refuge conducts periodic hunts to control this population. In the Florida Keys, Gambian giant pouched rats (*Cricetomys gambianus*) have established a localized, reproducing population and are the subject of a multi-agency eradication effort.

## INTRODUCTION PATHWAYS

Florida has had a long history of tourist attractions, many of which featured exotic animals, prior to the establishment of captive wildlife regulations. Several non-native species escaped from these facilities either from lax security or from defunct businesses that abandoned their operations. Florida aquaculture, though well regulated in recent years, has contributed to the release of freshwater non-native species. Anecdotal evidence suggests that a possibly established population of Argentine black and white tegu (*Tupinambis merianae*) may have originated when a dealer released his stock in response to falling market prices. Many non-native lizard observations are in proximity to reptile dealers.

In addition, Florida has three significant ports, and the Port of Miami receives shipments of tens of thousands of imported tropical fish, reptiles, amphibians and plants. Although most specimens have destinations outside Florida, many are consigned to local dealers and breeders.

With the ready availability of non-native pets, it is not surprising that released animals are a major source of introduction. For reptiles, prior to 1950, the few introductions occurred collaterally with freight shipments. Since that time, the volume of introductions has increased sharply, primarily through the pet trade (Figure 1).

Florida's tropical and subtropical climate further improve the likelihood of success for non-native species, particularly fishes and reptiles. Another aspect conducive to establishment is the relatively depauperate ichthyofauna and herpetofauna in much of peninsular Florida. Unlike most tropical areas of the world, Florida is isolated from land with similar climate. Most native species originated from temperate regions of the southeastern United States, many at the extreme southern end of their

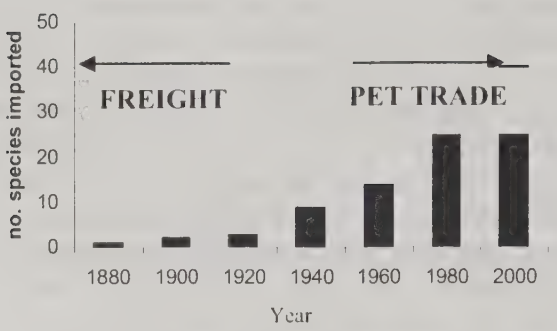



**Figure 1.** Introduction pathways for non-native reptiles and amphibians in Florida, 1880 - 2000. The Texas horned lizard (*Phrynosoma cornutum*) was imported by the pet trade in 1934, and is the sole exception to the freight pathway prior to 1940.

range. As a result, introduced species find a less species, including one native anole, whereas Cuba has 83 native lizard species, including 54 anoles. South Florida only has 10 native lizards and over three times as many exotic lizard species (K. Enge, FWC, personal communication). Of the 175 species of freshwater fish, many are cyprinids and darters restricted to the temperate streams in north Florida.

Cultural influence appears to have opened new pathways, particularly for non-native fishes. Asian food markets frequently sell live fish, and the bullseye snakehead (*Canna marulius*), and possibly the swamp eel (*Monopterus alba*) originated from such operations. Live northern snakeheads (*C. argus*), a prohibited species, have been confiscated by FWC law enforcement. Brown hoplo (*Hoplosternum littorale*), a small South American catfish prized as food item, may have been spread by immigrants with no knowledge of laws to the contrary. A Venezuelan immigrant confided to an FWC biologist that he regularly placed hoplo in southeast Florida canals to have a readily available food source.

## FWC HISTORY

In response to a growing number of introduced freshwater fishes, biological investigations commenced in the early 1960s, and the Non-Native Fish Research Laboratory was established in south Florida in 1966. Based on investigations by Lab staff, in 1976 several freshwater fishes were legally designated for restricted possession, i.e., these could be possessed only under strict biosecurity by research institutions and public exhibitors, and, under more limited circumstances, commercial producers. This list was re-evaluated in the early 1980s when several species were designated as prohibited, removing the possibility of commercial use. These regulations have been modified periodically, including the addition of aquatic organisms other than fishes, zebra mussel (*Dreissena polymorpha*) in 1993, Australian red claw crayfish (*Cherax quadricarinatus*) in 1996, and Chinese mitten crab (*Eriocheir sinensis*) in 1999, the latter at the suggestion of the aquaculture industry.

The first regulations for the possession of captive wildlife, primarily exotic species, were established in 1970. This was followed by the creation of a specialized law enforcement unit staffed by officers with degrees in biology to inspect facilities and survey ports to intercept restricted and prohibited imports. Florida's captive wildlife regulations, the most comprehensive in the United States, continue to evolve. In 2005, a comprehensive review of the regulations was undertaken in conjunction with stakeholders from affected industries and organizations (Table 1), and the first of a series of modified regulations was approved by FWC in 2006.

In 2004, in response to stakeholder concerns, FWC created a section to provide a seamless and comprehensive approach to non-native species issues. This alignment will increase the agency's capability to respond to terrestrial non-native species. Additional emphasis will be placed on interagency coordination at the local, state and federal level. This cooperative approach is evident in the management examples cited below.

## MANAGEMENT APPROACH

The primary tenet in management of non-native species is prevention. There are three aspects of prevention: regulation, outreach, and surveillance.

### Regulation

FWC's position toward prevention is reflected in its primary non-native species regulation, which states that it is unlawful to "to possess, transport or otherwise bring into the state or to release or introduce in the state any freshwater fish, aquatic invertebrate, marine plant, marine animal, or wild animal life that is not native to the state unless such person shall first secure a permit from the Commission" (Rule 68-5.001, Florida Administrative Code). Violation of this rule is a first degree misdemeanor, punishable by a fine up

**Table 1.** Captive wildlife technical assistance representation, Florida Fish and Wildlife Conservation Commission, 2007.

| Representative | Organization | Interest |
|---|---|---|
| Alexa Strauss | FELD Entertainment, Inc. Ringling Brothers and Barnum & Bailey Circus | Entertainment |
| Terri Parrot-Nenezian, DVM | | Wildlife Rehabilitator |
| Dan Martinelli | Treasure Coast Wildlife Hospital | Wildlife Rehabilitator |
| Eugene Bessette | Ophiological Services, Inc. | Pet Industry |
| Joe Christman | Disney's Animal Kingdom | Entertainment |
| Leroy Coffman, DVM | Consultant | Pet Industry |
| Ken Johnson | Humane Society of the US | Animal Welfare |
| Susan Clubb, DVM | Hurricane Aviaries Parrot Jungle Island | Pet Industry and Entertainment |
| Bill Armstrong | Hillsborough County Animal Control Florida Animal Control Assoc. | Animal Control |
| R. Donavan Smith | Close Up Creatures, Inc. NGALA Private Reserve | Pet Industry |
| Kathy Stearns | Stearns Zoological Rescue and Rehabilitation | Wildlife Rehabilitator |

to $1,000 and imprisonment for as long as one year. However, this general proscription against unauthorized release does not prevent the possession and display of non-native species, which is addressed through risk-based captive wildlife and non-native species regulations.

Captive wildlife regulations deal primarily with species maintained as personal pets and for public exhibition. There are three classes of captive wildlife, based on the potential danger to human safety, each with permitting requirements (Table 2). Caging and confinement requirements reflect concerns over safety as well as for the welfare of the animal.

Prior to 2007, permits were not required to possess non-venomous, non-native reptiles. In response to the established Burmese python and Nile monitor populations, effective January 1, 2008, a $100 permit will be required to possess five species of large constrictor and one large lizard. Additionally, all specimens of these "reptiles of concern" must be permanently identified with a passive integrated transponder tag. These regulations are designed to discourage impulse purchases, and subsequent illegal release, that may lead to establishment of these large species.

Non-native species regulations originated from freshwater fish investigations and list species as 'conditional' or 'prohibited' based on risks posed to native fish and wildlife, economic impacts, or human health and safety (Table 3). In 2007, two species of personal pets were added to the list, based on ecological and economic risks. The red-eared slider was designated as conditional due to its intergradation with the native yellow-bellied slider (*T. s. scripta*). African giant pouched rats (*Cricetomys* spp.) were designated as prohibited based on risks to Florida agriculture, native listed rodent species, and to human health.

**Outreach**

Legal restrictions serve as guidance for owners of non-native animals but are ineffective for those unaware of regulations, or individuals who do not understand their rationale. Many residents have moved only recently to Florida, and have no appreciation of native species and habitats. Accordingly, outreach has substantial preventive value and FWC has endeavored to more fully embrace this management tool. The FWC website has a second level page dedicated to non-native species (www.myfwc.com/nonnatives).

**Table 2.** Regulatory classes of wildlife in Florida.

| **Species/Group** | **Class I** (for exhibition only; no personal possession, except for those owned prior to 1980; $250 permit fee) | **Class II** (permit fee $140; applicant must demonstrate one year or 1000 hours experience for each species possessed) | **Class III** (no cost permit; applicant must be at least 16 years old) Other Wildlife not classified as I or II or exempt fall into this class. |
|---|---|---|---|
| Primates | Chimpanzees (*Pan*), Gorillas (*Gorilla*), Gibbons (*Hylobates*), Drills and mandrills (*Mandrillus*), Orangutans (*Pongo*), Baboons (*Papaio*), Siamangs (*Symphalangus*), Gelada baboons (*Theropithecus*) | Howler monkeys (*Alouatta*), Uakaris (*Cacajao*), Mangabeys (*Cercocebus*), Guenons (*Ceropithecus*), Bearded sakis (*Chiropotes*), Guereza monkeys (*Colobus*), Celebes black apes (*Cynopithecus*), Idris (*Indri*), Macaques (*Macaca*), Langurs (*Presbytis*), Douc langurs (*Pygathrix*), Snub-nosed langurs | |
| Felids | Snow leopards (*Panthera uncia*), Leopards (*Panthera pardus*), Jaguars (*Panthera onca*), Tigers (*Panthera tigris*), Lions (*Panthera leo*) | Servals (*Leptailurus serval*), European and Canadian lynx (*Lynx lynx*), Cougars & panthers (*Puma concolor*), Bobcats (*L. rufus*), Cheetahs | |
| Canids | | Coyotes (*Canis latrans*), Gray wolves (*C. lupus*), Red wolves (*C. niger*), Asiatic jackals (*C. aureus*), Black-backed jackals (*C. mesomelas*), Side-striped jackals (*C. adustus*), Indian dholes (*Cuon alpinus*), African hunting dogs (*Lycaon pictus*) | |
| Mustelids | | Wolverines (*Gulo gulo*), Honey badgers | |
| Other | Bears(Ursidae), Rhinoceros (Rhinocerotidae), Elephants (Elephantidae) Hippopotamuses (Hippopotamidae), Cape buffalos (*Syncerus caffer caffer*) | Binturongs (*Arctictis binturong*), Hyenas (Hyaenidae) | |
| Reptiles | Crocodiles, except dwarf and Congo (Crocodilidae), Gavials (Gavialidae), Black caimans (*Melanosuchus niger*), Komodo dragons (*Varanus komodoensis*) | Dwarf crocodiles (*Osteolaemus tetraspis*), Alligators, caimans (except American alligator), (Alligatoridae) | |
| Rattites | | Ostrich (*Struthio camelus*), Cassowary (*Casuarius*) | |

**Table 3.** Non-native species with restrictions on possession in Florida. Refer to regulations in the Florida Administrative Code for more details.

| | |
|---|---|
| **Conditional Species** (may be possessed for the purpose of public exhibition, research, or commercial purposes) | |
| Freshwater fishes | Grass carp (*Ctenopharyngodon idella*), Bighead carp (*Aristichthys nobilis*), Silver carp (*Hypophthalmichthys molitrix*), Snail or black carp (*Mylopharyngodon piceus*), Dorados (*Salminus* spp.), Nile perches (*Lates* spp.), Blue tilapia (*Oreochromis aureus*), Wami tilapia (*O. hornorum*), Mozambique tilapia (*O. mossambicus*), Nile tilapia (*O. niloticus*), Walking catfish (*Clarias batrachus*), Blue catfish (*Ictalurus furcatus*), Freshwater stingrays (*Potamotrygonidae*), and Bony-tongue fishes (*Osteoglossidae*) (except silver arowana [*Osteoglossum bicirrhosum*]) |
| Freshwater Invertebrates | Australian red claw crayfish (*Cherax quadricarinatus*), Red swamp crayfish (*Procambarus clarkii*), and White river crayfish (*Procambarus zonangulas*) |
| Reptiles | Red-eared slider (*Trachemys scripta elegans*) |
| Mammals | Nutria (*Myocastor coypu*) |
| **Prohibited Species** (may be possessed only by accredited public exhibitors or by research institutions with an approved research plan) | |
| Freshwater fishes | African electric catfishes (*Malapteruridae*), African tigerfishes (*Hydrocyninae*), Airbreathing catfishes (*Clariidae*) (except Walking catfish [*Clarias batrachus*]), Candiru catfishes (*Trichomycteridae*), Freshwater electric eels (*Electrophoridae*), Lampreys (*Petromyzonidae*), Piranhas and pirambebas (*Serrasalminae*), Snakeheads (*Channidae*), Tilapias (*Tilapia spp., Sarotherodon spp., Oreochromis spp.*) (except conditional *Oreochromis* species), Trahiras or tigerfishes (*Erythrinidae*), Airsac catfishes (*Heteropneustidae*), and Green sunfish (*Lepomis cyanellus*) |
| Freshwater Invertebrates | Australian crayfish (*Cherax* spp.), except *C. quadricarinatus* in a closed tank system, Zebra mussels (*Dreissena polymorpha*) |
| Mammals | African giant pouched rats (*Cricetomys* spp.) |
| Marine species | Mitten crabs (*Eriocheir*), Sea snakes (*Hydrophiidae*), Weeverfishes (*Trachinidae*), Stone fishes (*Synanceia* spp.) |

To close the personal pet release pathway, FWC has initiated a Non-native Pet Amnesty program. Conducted on weekends, these well-publicized events provide an opportunity to surrender non-native pets without penalty. The format calls for multi-agency involvement with substantial focus on outreach. Two events have been conducted with a third planned for February 2008 in Miami. A major concern of pet owners is the welfare of their animal, and arrangements are made for adopters to be at events to minimize the possibility that animals will be euthanized. All pets have been successfully adopted, with the exception of two specimens that were unhealthy at the time of surrender.

The volume of animals surrendered at one-day events is miniscule compared to the number of exotic pets in the state, and the principal value of the weekend Amnesty program is to increase awareness of the problem of non-native species in Florida. The decision to release a pet is based on the circumstances at hand, and we have no illusion that owners will postpone their choice to discard a non-native animal until an event occurs. FWC's ultimate objective is to develop a pet surrender network, with qualified adopters ready to accept animals on an ongoing basis. The network will be mediated through the website.

Other outreach efforts target specific audiences and leverage exposure. FWC developed a poster highlighting several non-native freshwater fishes, their native continent and their pathway to Florida. This will be distributed at county nature centers, museums, and science classrooms. FWC has pursued a cooperative, rather than adversarial, relationship with the pet industry, and agency attendance at trade shows provides an opportunity to efficiently reach out to dealers as well as potential pet owners.

Florida's diverse socioeconomic profile presents significant challenges for outreach and education. Extensive Cuban, Mexican, Haitian, and Vietnamese communities, among others, present language as well as cultural barriers. Simple translation of English-language media will be inadequate, and research is needed to determine effective methods to overcome these cultural challenges.

### Surveillance and Rapid Detection

When preventive measures fail, it is important to detect non-native species as soon as possible to improve the likelihood of eradication or containment. Florida has an interagency working group to coordinate efforts to prevent and manage non-native and invasive plants and animals. Recognizing that many agencies have staff in the field, Florida state agencies are attempting to utilize this resource to improve surveillance. An initial attempt to map the distribution of exotic applesnails (*Pomacea* spp.) by providing reporting forms to field staff had limited success, likely due to competition with internal priorities.

The vast majority of introduced species are reported by the public, frequently well after their initial discovery. Accordingly, FWC is in the initial stages of developing a web-based reporting system. The difficulty is in finding the balance of providing adequate information to limit bogus reporting while not overwhelming the non-professional. A simplified version has been employed to solicit reports on Gambian giant pouched rat sightings in the vicinity of their distribution in Grassy Key, Florida (myfwc.com/nonnatives/gambianRat.htm), using photographs to help the public distinguish similar species. This effort has not been widely publicized and has yielded few submissions, including some from areas hundreds of miles from the putative Florida distribution of this species.

Unlike the simple on-line reporting format, a more complex reporting process is in place for non-native applesnails, capitalizing on the visibility and distinctiveness of their eggs. The web document provides background on biology, potential impacts, control, and federal regulations, accompanied by photos of eggs and snails (myfwc.com/nonnatives/Docs/FWC_applesnails_FLMS_handout.pdf). The reporting process calls for the observer to complete a one-page form and submit via facsimile. We have not had an opportunity to assess the quality of reports to date. A reporting format for Reptiles of Concern is under development.

## RAPID RESPONSE

FWC's approach calls for elimination of non-native species populations where practical. The majority of introduced species will not become invasive. Regardless, FWC would prefer to eradicate incipient populations rather than risk ecological or economic damage, or harm to human health. Upon detection of a non-native species, a rapid assessment is recommended to determine if eradication is practical, i.e., within financial means, without unacceptable impacts to native species, and commensurate with the potential impact of the introduced organism.

Prior to 2004, several isolated non-native fishes were eradicated by rotenone poisoning of the water

body. Since 2004, four rapid assessments have been conducted to gauge the potential for eradication. In one instance a limited number of an exotic aquarium species, silver dollar (*Metynnis* spp., Characidae), were discovered in an isolated 40-ha lake, including evidence of reproduction. Based on the cost of applying rotenone to this water body, the presence of native predatory fish, and the small number of silver dollar encountered, a decision was made to continue sampling rather than pursue eradication. Subsequent observations indicate this introduced species has been eliminated by natural causes.

In 2004, a population of Gambian giant pouched rats in Grassy Key was reported to the U.S. Fish and Wildlife Service (USFWS). Following preliminary investigations by that agency and USDA APHIS Wildlife Services (WS), FWC organized a multi-agency planning effort in March 2005, which led to an initial abundance index conducted by WS, along with live trapping and camera surveys to delineate the range of the population. Funding for eradication was unavailable, so FWC obtained a $20,000 grant matched by WS to conduct a pilot eradication project on a portion of the range in June 2006. Subsequently, FWC and WS committed internal funds, along with a grant from USFWS, and in-kind contributions from the South Florida Water Management District (SFWMD) and the Florida Department of Environmental Protection, to initiate an eradication project in early 2007. The first full-scale operation to eliminate this population occurred over two years after the initial report. However, based on field observations, this did not decrease the likelihood of success. As of this writing, the first of several periodic surveys to assess survival of Gambian giant pouched rats was underway.

Purple swamphens were first observed in urban southeast Florida in 1996, likely having escaped from a local aviculturist. Over the next several years, this population increased to over 200 birds but remained confined to developed areas. In early 2006, SFWMD vegetation management contractors reported purple swamphens in the Everglades Conservation Areas. Subsequently, SFWMD staff observed these birds in some of their water treatment areas. In May 2006, FWC and SFWMD conducted a survey of the areas, which indicated a population of approximately 100 birds. In August 2006, night capture methods were evaluated and found to be ineffective. In October 2006, FWC and SFWMD initiated an eradication effort using shotguns and actively pursuing purple swamphens from airboats. As of March 2007, 601 purple swamphens had been killed, leaving a casual estimate of about 50 birds. Subsequent observations have been limited by extremely low water, but additional removal efforts were scheduled for summer 2007, and both agencies remain guardedly optimistic that, at the least, potential impact to native wildlife and vegetation can be minimized.

Argentine black and white tegus were reported from west central Florida in 2006 by Hillsborough County Environmental Services staff, as well as by private land managers in neighboring Polk County. Initial surveys in summer 2006 indicated usage by juvenile tegus of gopher tortoise (*Gopherus polyphemus*) burrows on public land, corroborating observations by consultants conducting habitat reclamation projects on disturbed lands. Posters were developed to solicit public observations in an attempt to delineate the range of this introduced lizard. Based on credible public reports, it appears that this tegu species occurs over at least 100 $km^2$, although its distribution and habitat preference are not well understood. A member of the Teidae family, tegus were presumed to enter hibernation in late summer, delaying further sampling or removal. A lack of funding and staffing have hampered further efforts at gathering the data requisite to determining if eradication is feasible, and FWC is not optimistic that this species will be eliminated.

## ASSESSMENT

In the event that extirpation of an introduced species is deemed to be impractical, assessment of its ecological role is in order to determine appropriate management. Among the factors to be considered are age and growth, diet, reproduction, limiting factors (e.g., temperature, habitat, salinity), and population abundance and composition of introduced and native species. These data may be used to estimate impact, and, consequently the appropriate level of intervention. At the outset, intensity of these studies is high, followed by a reduced level of effort over an extended period.

Most FWC assessments have been of freshwater exotic fishes, focused primarily in the canals of southeast Florida where the greatest number of introductions have occurred. Studies over the past three decades have documented no impacts to native fishes exclusively attributed to introduced species, contrary to our original suppositions. Similar longitudinal studies have not been conducted for non-native terrestrial wildlife, and

FWC will be pursuing research in this area to determine if the response is similar to our observations from the aquatic environment.

## CONCLUSION

Fortunately, very few of Florida's established non-native species have resulted in significant adverse ecological or economic impacts. However, the minority of problematic introductions can exact substantial ecological or economic damage and it is incumbent upon natural resource agencies to take all reasonable measures to discourage release and escape of non-indigenous fish and wildlife. FWC has chosen to encourage responsible pet ownership rather than adopt a prohibitionist approach, which they believe would be ineffective with the substantial level of pet ownership and the industry that services this demand.

There is a suggestion that the pace of vertebrate introductions or establishment in Florida is slowing in the current decade following an increase during the 1990s (Figure 2). This may be due to increased agency focus at the state and national level. However, the pattern of explosive population growth, including diverse cultural influences will continue to be a challenge to natural resource managers. The most effective barrier to the introduction of non-native organisms is to identify species that pose the greatest risks and limit their possession. Since the development of FWC's non-native species rules, only two fishes have become permanently established, along with two possibly established species. One of the established species, spotted tilapia (*Tilapia mariea*) may have been present yet not observed at the time of its listing. Better screening (i.e., risk analyses) are needed to aid in this process, and this should be a focus of research.

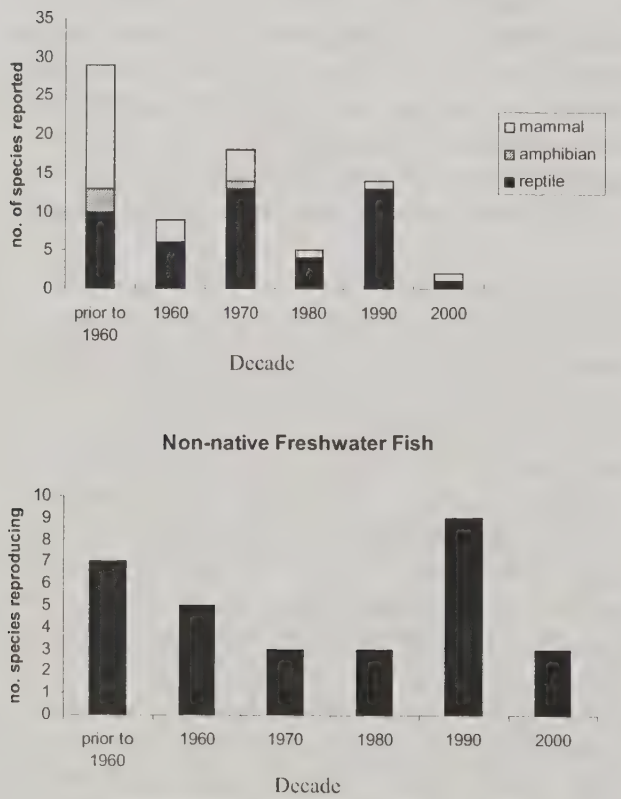



**Figure 2.** Introduction of non-native vertebrates in Florida by decade.

## LITERATURE CITED


BOMFORD, M., F. KRAUS, M. BRAYSHER, L. WALTER, AND L. BROWN. 2005. Risk assessment model for the import and keeping of exotic reptiles and amphibians. Report to the Australia Bureau of Rural Sciences.

SHAFLAND, P. L., K. B. GESTRING, AND M. S. STANFORD. In review. Criteria for categorizing introduced fishes collected from open waters. Fisheries.

WILLIAMSON, N. AND A. FITTER. 1996. The varying success of invaders. Advances in Invasion Biology 77:1651-1666.

# CUSTOM TRUCKS, RADIO SNAKE JINGLES, AND TEMPORARY TATTOOS: AN OVERVIEW OF A SUCCESSFUL PUBLIC AWARENESS CAMPAIGN RELATED TO BROWN TREESNAKES IN THE COMMONWEALTH OF THE NORTHERN MARIANA ISLANDS

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***Abstract:*** The brown treesnake (*Boiga irregularis*) was introduced on Guam during post World War II cargo movements. Brown treesnakes (BTS) have now become exceptionally abundant on Guam and pose a direct, significant, and growing threat to other areas outside of their historic range, including the Commonwealth of the Northern Mariana Islands (CNMI), the State of Hawaii, the United States mainland and other sites regionally and internationally. The CNMI is at high risk for a BTS introduction due to its close proximity and the types and amount of cargo received. Limited BTS awareness efforts between 1986 and 2002 in the CNMI resulted in an average response time of 126 hours indicating that an increased awareness of appropriate response actions among the public was needed. An awareness campaign was outlined in three phases: (1) baseline public survey, (2) ten month awareness campaign, and (3) re-evaluation survey. The CNMI Division of Fish and Wildlife (DFW) staff and an outside advertising firm created a baseline survey to gather media consumption information and environmental attitudes. Conclusions of the baseline survey were then utilized to create a ten month awareness campaign to improve average response times. After the campaign, average response times decreased from 126 hours to 1 hour 42 minutes. The re-evaluation survey confirmed the success of specific campaign components and directs current awareness efforts.

***Key Words:*** *Boiga irregularis*, brown treesnake, education, invasive species, public awareness, rapid response.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K.A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Shortly after World War II and before 1952, the brown treesnake (*Boiga irregularis*) was accidentally transported from its native range to Guam, probably as a stowaway in ship cargo (Rodda et al. 1992). Upon arrival in Guam, the brown treesnake (BTS) encountered an abundant prey base and an absence of natural predators and pathogens allowing the BTS to reach high densities (Savidge 1987). Since its accidental introduction to Guam, the BTS has been responsible for the extirpation of ten native forest bird species; three native lizard species and hundreds of power outages affecting private, commercial, and military activities; large-scale loss of domestic birds and pets; numerous potentially fatal envenomations of children; and considerable emotional trauma to residents and visitors alike (Fritts et al. 2001, Campbell 2004).

## HIGH-RISK EXTRALIMITAL SITES

Guam is located in the western Pacific and serves as a military and commercial shipping hub for the region. Due to its high abundance on Guam, the BTS poses a direct, significant, and growing threat of dispersal to other areas outside of its historic range, including the Commonwealth of the Northern Mariana Islands (CNMI), the State of Hawaii, the mainland of the United States (US), and other regional and international sites (Campbell 2004). Unlike Guam, where the BTS has devastated many of the native wildlife populations, the CNMI still harbors many native species. Since 1982, however, the CNMI has documented 13 BTS captures (11 on Saipan and 2 on Rota) and 76 credible sightings. Repeated sightings on Saipan indicate that an incipient population is now present there (Colvin et al. 2005).

## PREVIOUS BROWN TREESNAKE AWARENESS EFFORTS IN THE CNMI

Rapid response and early detection are arguably the best practices in locating and eradicating invasive species. However, the CNMI rapid response efforts to potential BTS sightings have suffered from a lack of public awareness. From 1992 to 2002, BTS awareness efforts in the CNMI were limited primarily to school presentations, newspaper advertisements, and press releases. Consequently, the average response time (time elapsed from initial sighting to an official response by CNMI Division of Fish and Wildlife [DFW] personnel) for CNMI BTS sighting reports from August 1986 to August 2002 was approximately 126 hours (5.25 days). On November 15, 2001, the DFW received a telephone call from a professor of biology at the Northern Marianas College, Saipan, regarding a snake sighting that occurred on November 1, 2001. If early detection and eradication efforts were to succeed, DFW would have to improve the average response time by promoting public awareness.

## CNMI BTS AWARENESS CAMPAIGN DESIGN

The United States Department of the Interior's (DOI) Office of Insular Affairs (OIA), who had been funding the CNMI BTS Program since 1992, awarded an awareness grant to the DFW in January 2002 (Table 1).

**Table 1.** CNMI BTS Awareness Campaign Budget (2002-2007).

| | |
|---|---|
| Campaign awards 2002-2004: | |
| DOI-OIA grant | $100k |
| Telecommunication (gifts-in-kind) | $33k |
| Total award | $133k |
| Campaign award 2004-2007: | |
| USFWS | $65k |
| Campaign expenses 2002-2004: | |
| Baseline survey | $55k |
| 10-month campaign | $64k |
| Re-evaluation survey | $14k |
| Campaign expense 2004-2007: | |
| 36-month campaign | $65k |

The performance measurement of the OIA grant was to ultimately increase public awareness efforts to improve the average response times for snake sightings in the CNMI. Working closely with United States Geological Survey (USGS), DFW outlined a comprehensive awareness campaign that included three major components: (1) a baseline survey, (2) campaign design and implementation based on survey results, and (3) a re-evaluation survey that would measure the performance of the campaign. After outlining the awareness campaign, DFW contracted a local advertising firm to facilitate and expedite the campaign.

## BASELINE SURVEY

DFW worked closely with the advertising firm to develop a baseline survey that would generate information accurate to within ± 5% at the 95% confidence level. The survey was designed to measure current BTS awareness levels and the impact of past communications programs. Due to DFW planned implementation of BTS specific quarantine regulations, the survey also assessed viewpoints towards BTS specific regulations, as they would likely increase consumer costs. Because of the potential political impact of BTS specific quarantine regulations, the November 2002 survey was based on a random sample of Saipan's (CNMI) eligible voters.

The baseline survey revealed that the BTS problem had become a salient issue in the CNMI. Over 69% of the 300 people interviewed perceived the BTS to be "a problem", with 37% of the 69% perceiving the BTS as a "big problem." The survey also found that when asked if the quarantine was "a good move" even if it "results in delays" and "forces up costs", 53% of voters responded positively. When asked even if the quarantine list included "household items" shipped to Saipan, voter support remained high at 62% (Merrill 2004).

The baseline survey also provided information on media consumption habits in the CNMI. The survey found radio advertisements to be as equally effective in reaching audiences as more expensive newspaper and television advertisements. The media consumption information collected from the baseline survey was invaluable, as it had not been evaluated by a government agency in the past and was therefore unavailable.

## IMPLEMENTATION OF THE CNMI BTS AWARENESS CAMPAIGN

The first step in the implementation of the ten-month campaign was to develop clear and concise messages regarding appropriate community response actions to a snake sighting. It was evident

from the baseline survey that the public was generally aware of the BTS threat to the CNMI and that quarantine measures would be supported. However, the survey revealed confusion when the public was asked who they should contact to report BTS sightings in the CNMI. Since the objective of the campaign was to improve average response time, the messages needed to stress the immediacy in reporting any snake sightings to a BTS Hotline.

The "Don't Give Snakes a Break" slogan, accompanied with a logo and radio jingle, was created to highlight the hotline and stress the immediacy of reporting snake sightings. Advertisements were also developed to outline the threats of the BTS to the CNMI. The materials developed were incorporated in all public relations activities.

DFW contacted the local telecommunications company to establish the BTS Hotline. The telecommunications company immediately became interested in the campaign and offered to contribute cash donations, free advertisements, billing inserts and cellular phones to help improve BTS awareness in the CNMI. The partnership was formed and the 28-SNAKE Hotline (287-6253) was implemented in April 2003 during a press conference at the CNMI Governor's Office.

Over the course of the ten-month campaign, several outreach materials were created and distributed. To promote of the new hotline and partnership with the telecommunications company, 28-SNAKE t-shirts, hats, can coozies, temporary tattoos, and bumper stickers were made available. Radio advertisements, in the form of snake jingles, were aired during peak listening times. Radio contests with prizes donated by the telecommunications company were also held to test the knowledge of listeners. DFW also contracted a local artist to design custom paint jobs for the DFW trucks that acted as moving billboards for the BTS program. The trucks were entered in car shows, parades and parked at local events to promote the hotline.

To prepare for the 2003-04 school year, DFW created BTS book covers, brochures and stickers. In August, just as the school year began, the BTS campaign launched a series of advertisements to encourage educators to schedule BTS awareness presentations. During the year DFW staff averaged three presentations per week to elementary, middle and high school students.

## RE-EVALUTION SURVEY

In March 2004, DFW conducted a survey to evaluate the effectiveness of the BTS awareness campaign. A total of 356 individuals were surveyed with an accuracy of ± 5% at the 95% confidence level. Unfortunately, the 2002 survey (CNMI population of eligible voters) over-represented Chamorros and Caucasians and under-represented other ethnic groups. Therefore, the 2004 survey expanded to populations that included non-resident workers of various ethnic backgrounds in order to include more individuals in professions (night security, port equipment operators, construction, shipping and receiving) likely to encounter the BTS.

Some key demographic statistics (age, sex, number of dependents and level of education) remained constant between the two surveys. The re-evaluation survey, however, sampled a more evenly-distributed ethic population and found 46% of the respondents had incomes lower than $20,000 as compared with 28% in 2002 (Merrill 2004). It was known that the ethnic shift between the two surveys might represent a decrease in exposure to the campaign, as many of these groups were not the target audiences. However, the information from these groups would need to be known before any adjustments to the campaign could be made.

The BTS issue remained an important concern among those surveyed in 2004. In fact, the re-evaluation survey found that those perceiving snakes as a "very big problem" increased from 37% in 2002 to 57% in 2004. The 2004 survey also evaluated the messaging of the campaign and found 72% of the respondents were able to identify the correct slogan and hotline number. The survey also found 72% of the respondents were aware of the campaign's messaging regarding the snake's impact on wildlife, human health, and the economy (power outages) (Merrill 2004).

Even though the 2004 survey expanded to non-target audiences, the exposure percentage remained constant at 39% (both surveys) having seen a BTS presentation. A campaign exposure index (CEI) was calculated from the two surveys to find that 87% (2002) and 80% (2004) of the respondents had seen at least two or more types of BTS awareness messaging. The survey also measured the effectiveness of specific campaign outreach materials. Results showed custom trucks and bumper stickers had the highest exposure values. The CEI found that between ethnic groups, Chamorros (local population) held the highest

scores in both 2002 and 2004 with 37% and 38%, respectively. Filipinos had relatively high scores as well with 15% and 23%, respectively, and other Asians, 8% and 14%, respectively (Merrill 2004).

An important component of the 2002 and 2004 surveys was to determine the level of public support for planned BTS-specific quarantine regulations. The surveys found that nearly 9 out of 10 respondents (87% in 2002; 85% in 2004) agreed that increased quarantine was a "good move." Furthermore, when asked if quarantine was still a "good move" if it resulted in delays and increased costs, 53% of respondents in 2002 and 73% in 2004 said that it was (Merrill 2004).

A quarantine support index (QSI) was calculated from both surveys to measure the respondents' incremental support for various quarantine options. This was used to gauge the respondent's overall support. It was evident that exposure to the BTS awareness campaign was associated with support for quarantine. Respondents in both surveys scoring the highest on the CEI also scored highest on the QSI (49% November 2002; 57% March 2004). Additionally, the surveys revealed that respondents with lower exposure tended to have more mixed (or ambiguous) views about the planned quarantine regulations (Merrill 2004).

## CURRENT BTS AWARENESS EFFORTS IN THE CNMI

After the 2004 survey, the DFW received an awareness grant from the US Fish and Wildlife Service (USFWS) to continue the campaign. Utilizing the information from previous efforts, DFW continued to expand the BTS awareness campaign to target additional audiences. In June 2004, the DFW expanded its presentation activities by coordinating with non-resident worker groups and presenting awareness information to over 3,000 non-resident workers by June 2007. The DFW also increased awareness efforts with the public school system by incorporating BTS information and presentations into the 4$^{th}$ grade science curriculum. Furthermore, DFW expanded awareness efforts and helped to offset budget shortfalls in the public school system by providing basketball backboards with BTS messaging. DFW plans to further this effort by renovating school bus stops and sports fields.

Additional outreach materials were developed including, banners, revised snake jingles, mouse pads, "push" and "pull" stickers for places of business and e-advertisements for visiting military personnel.

## DISCUSSION

The success of the CNMI BTS awareness campaign can ultimately be measured by the improved average response time from 126 hours (06/1986-06/2002) to 1 hour 42 minutes (08/2002-present). This was achieved by developing appropriate motivational messages and materials that were based upon existing audience behaviors and attitudes. The baseline and re-evaluation surveys proved to be invaluable for conducting the BTS awareness campaign. Collecting public attitude and media information before and after a campaign allows mangers to evaluate the effectiveness of the campaign.

## LITERATURE CITED


CAMPBELL, E. W., III. 2004. Brown treesnake fact sheet, unpublished report.

COLVIN, B. A., M. W. FALL, L. A. FITZGERALD, L. L. LOOPE. 2005. Review of brown treesnake problems and control programs, Report of Observations and Recommendations, unpublished report.

FRITTS, T. H., AND D. LEASMAN-TANNER. 2001. The Brown treesnake on Guam: how the arrival of one invasive species damaged the ecology, commerce, electrical systems, and human health on Guam: a comprehensive information source. Available Online: http://www.fort.usgs.gov/resources/education/bts/bts_home.asp.

MERRILL, J. 2004 Follow-up survey: CNMI residential attitudes of the brown treesnake, unpublished report.

RODDA, G. H., T. H. FRITTS, AND P. J. CONRY. 1992. Origin and population growth of the brown treesnake (*Boiga irregularis*) on Guam. Pacific Science 46:46-57.

SAVIDGE, J. A. 1987. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660-668.

# PROMOTING AWARENESS, KNOWLEDGE AND GOOD INTENTIONS

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***Abstract:*** Two independent studies in the 1990s found that Hawaii had the nation's worst alien pest problem due to gaps in prevention, detection and control programs, which could be addressed through increased communication, cooperation, and public outreach. The Coordinating Group on Alien Pest Species (CGAPS), a government/non-government partnership, formed in 1995 to address these gaps and work on public awareness. Using snakes as an example, a 1996 Hawaii statewide public awareness survey found that 66% of people "had heard of brown treesnakes." CGAPS ran the "Silent Invasion" campaign in 1997, with "shock footage" television commercials and specials about brown treesnakes on Guam. By 2004, awareness of brown treesnakes had risen to 83%, and 91% of Hawaiian residents were "very likely" to report snake sightings, yet less than 5% knew about the Pest Hotline, nobody could recite it, and some balked at having to call an Oahu number to report a snake. CGAPS launched phase two of the campaign in 2006, with television and print media and a new toll-free hotline number. Follow-up surveys confirmed rising awareness about brown treesnakes, but unacceptably low awareness and use of the Pest Hotline. CGAPS is experimenting with ways to promote knowledge and good intentions to different audiences in Hawaii.

***Key Words:*** *Boiga irregularis*, brown treesnake, invasive species, media campaign, outreach, public awareness, snakes.

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## INTRODUCTION

For millions of years, the natural isolation of the Hawaiian Islands has been an effective barrier for preventing the arrival and colonization of many species, including all land reptiles, amphibians, and all but one land mammal, the Hawaiian hoary bat (*Lasiurus cinereus semotus*). The first non-native species arrived with Polynesian voyagers in about 400 A.D. Following Western contact by Captain Cook in 1778, an influx of alien species began that has continued through today (Stone 1985).

The first laws and programs aimed at protecting Hawaii from invasive species were set in place in the late 1800's, but many times these efforts were designed to address individual pests, or protect the interests of big agriculture industries, including sugar and pineapple. The result was a patchwork system of laws, agency authorities and mandates. In the 1990's, two studies reported that Hawaii had the worst alien pest problem in the nation due to large gaps in Hawaii's invasive species programs and laws, a lack of inter- and intra-agency communication, and cooperation, and concern (TNC/NRDC 1992, U.S. OTA 1993). The reports concluded that these gaps could be addressed through increased inter-and intra-agency communication, cooperation, and public outreach. In response to this, the Coordinating Group on Alien Pest Species (CGAPS), a voluntary government/non-government partnership, formed in 1995 to address these gaps and has made some headway on public awareness.

The overall goal of CGAPS outreach efforts is to raise awareness in Hawaii residents about invasive species in order to foster a sense of concern and result in supportive actions. Since 1996, CGAPS has conducted two television media campaigns, titled the Silent Invasion, and has taken advantage of outreach opportunities via the news media. One measure of outreach efficacy has been to conduct periodic telephone surveys of a representative number of residents statewide, to gain a sense of public awareness, concern, and support.

One outreach topic CGAPS focuses on is brown treesnakes (*Boiga irregularis*), and the threat that these and other snakes pose to Hawaii. The key outreatch messages include: (1) snakes are bad for Hawaii, (2) snakes are illegal to import or own, (3) if you see a snake, report it to the Hawaii Department of Agriculture (HDOA) pest hotline immediately to report it, and (4) people with illegal

pet snakes should take advantage of the amnesty program by calling the pest hotline to surrender snakes without fear of prosecution. This paper will look at CGAPS outreach efforts and public awareness of brown treesnakes and other snakes between 1996 and 2007.

## METHODS AND RESULTS

In October 1996, CGAPS contracted with a company to conduct the first baseline public awareness survey specifically on invasive species. Four hundred and seven registered voters were interviewed by telephone in a random sample, balanced according to all known demographic factors, for a margin of error of +/- 4.8% at a 95% confidence level. From these results, CGAPS learned that just 29% of people polled had heard of the concept of "alien pest species," yet 66% of people "had heard of brown treesnakes," (Kitchens Group 1996). Participants were not asked what number they would call if they saw a snake.

CGAPS ran the first Silent Invasion media campaign in 1997-98, which featured "shock footage" television public service announcements (PSAs), short topic specials and a 26-page booklet about brown treesnakes and other alien pest species. Although there was no follow-up survey after the campaign, public outreach continued via opportunistic news media stories through 2003, when outreach messages and needs were reevaluated in preparation for the second Silent Invasion campaign.

CGAPS contracted with a company to conduct three focus groups in October and November 2003, to evaluate and review campaign messages. Results showed that invasive species were not a "top of mind" concern, yet participants were aware of the brown treesnake and talked about the damage it caused on Guam. When it came to talking about reporting snakes, many participants said they would report them, yet some of the same participants relayed anecdotes about people they knew who had smuggled pet snakes into Hawaii, and yet they did not report them to authorities (Ward Research 2003). From these results, CGAPS concluded that it was important to continue to link snake issues with the brown treesnake story, and that previous messages regarding snakes were on target.

Focus groups also revealed some major issues with the pest hotline. Participants were not aware that the state had a pest hotline number, despite it having been in operation since 1992. Most participants could not identify what agency was responsible for the hotline. When participants were told the hotline was an Oahu number, neighbor island residents said they would be reluctant to call because they would incur long distance charges. Neighbor island residents also preferred to speak to someone on their island that might be familiar with the sighting locations (Ward Research 2003).

To address these hotline issues, a new statewide pest hotline number was set up in Hawaii to allow callers to dial the seven-digit number, 643-PEST, direct and toll free from any island. The new hotline number uses a computer program to automatically route calls to the nearest Hawaii Department of Agriculture (HDOA) office during work hours and to the HDOA office at the Honolulu International Airport after hours and on weekends, for 20 hours per day, 7-day coverage.

Following the establishment of a new toll-free hotline, a new public awareness baseline survey was conducted in November 2004 (Ward Research), prior to the second Silent Invasion media campaign, between January 2006 and May 2007. Two benchmark surveys were conducted during the campaign, in June 2006 (Ward Research), and February 2007 (Qmark Research and Polling), to gauge changes in awareness, detect problem areas, and allow for changes in the messages and delivery methods. These surveys again followed the same methodology, but with a slightly higher number of people surveyed (n=501) to maintain a sample size with a maximum sampling error of +/-4.4% at the 95% confidence level.

For the first phase of the new Silent Invasion campaign, CGAPS produced five, 30-second public service announcements (PSAs) which aired on all major television network affiliate stations from January through July, 2006. The snake PSA ran more than 200 times during high-viewership programs, such as the evening news, CSI Miami, Lost, Dancing with the Stars, and on local programs, such as Outside Hawaii, a show that appeals to hunters and hikers.

After airing the PSA for five months, the June 2006 benchmark survey measured an increase in the number of people that said that invasive species were "a very serious problem," from 36% prior to the campaign to 43% in 2006. In addition, public awareness of brown treesnakes started at 83% in 2004, rose to 88% during the airing of the PSA in 2006, and decreased slightly to 84% in 2007, after the five months without being broadcast (note again that the maximum sampling error of +/-4.4%, therefore the decline from 88% to 84% is not

considered statistically significant, although it warranted consideration).

When respondents were asked, "What number would you call if you saw a snake," people continued to have difficulty reciting the pest hotline number or identifying the appropriate agency. In the 2004 baseline survey, 5% of respondents could recite the Oahu-based number pest hotline number (the new 643-PEST number was instituted in December 2005). In the 2006 benchmark survey, none of the respondents could recite 643-PEST as the number to call, and 1% of respondents recited the old (but still operational) hotline number.

This prompted a switch in outreach methods from primarily television PSAs to a combination of television and radio between January and May, 2007, aimed primarily at teaching people the pest hotline number.

Popular local comedian Frank DeLima was hired to write and sing a pest hotline jingle, which was aired more than 3,500 times on radio stations statewide from January through April, 2007. During this same period, the snake PSA was aired again on three major television network affiliates, to maximize snakes as a "top of mind" issue. The February 2007 benchmark survey showed a modest (6%) increase in respondents able to recite the new hotline number (Qmark Research and Polling 2007).

Survey trends show that CGAPS has had some success in reaching the public with traditional media messages delivered by print, television and radio media, as can be seen by the rise in the number of people that "have heard of invasive species" (Figure 1), and brown treesnakes (Figure 2). It is also notable that more than 80% of people are "highly likely" to report a snake (Figure 3), despite the fact that the majority of Hawaii residents have no firsthand experience with snakes.

However, there remains a growing concern that some demographic groups were missed or not accurately targeted. For example, survey data from 2006 revealed that fewer 18-34 year olds think that invasive species are a very serious or somewhat serious issue (77%), compared to 82% of 35-54 year olds and 87% of those 55 and older (Figure 4). The 2006 survey also showed that fewer 18-34 year old males were likely to report a snake if they saw one.

CGAPS conducted a limited experiment using a local marketing company that specializes in marketing to 18-34 year olds. This targeted outreach project was aimed primarily 18-34 year old males, and secondarily at the entire 18-34 year old demographic, and ran from January through March 2007.

Although information about invasive species is readily available, there appears to be little reason for most 18-34 year olds to seek it out. For example, 38% of 18-34 year olds reported getting information from reading the newspaper, compared to 63% of 35-54 year olds and 68% of those 55 and older (Qmark Research and Polling 2007). Therefore, it was decided that campaign materials must create a compelling reason for the 18-34 year old target audience to seek information and provide the information in a way that might appeal to their generation.

Using principles made popular in the book, "The Tipping Point: how little things can make a big difference," by Malcolm Gladwell (2000), a trial campaign was designed to capitalize on the potential ability of selected socially popular women to connect with their social networks to promote CGAPS' standard snake messages. A new website (643-PEST.com) was set up to look like a horror movie trailer, featuring six alluring women "starring" as six invasive species, including a

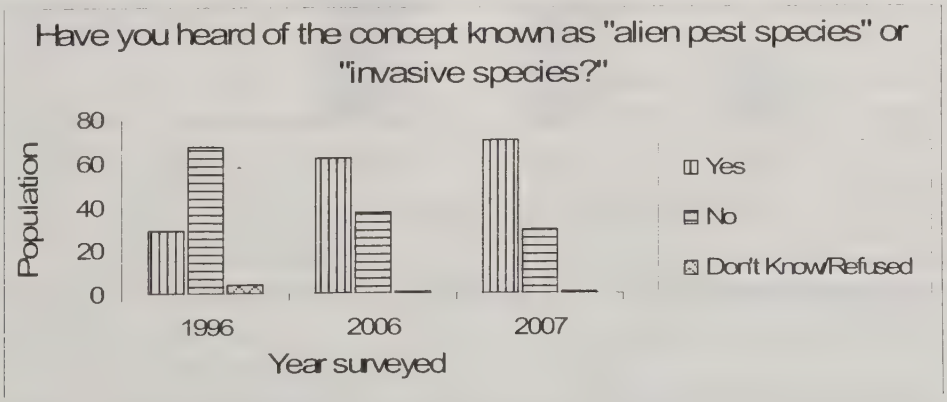



**Figure 1.** The number of respondents that have heard of the concept of invasive species has risen steadily since outreach efforts began in 1997.

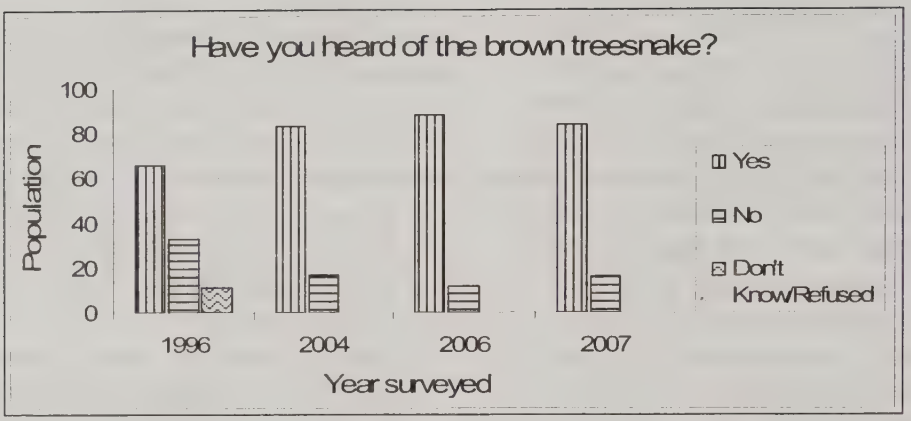



**Figure 2.** The number of respondents that have heard of brown treesnakes has also risen over the years, with a peak occurring in 2006 during the airing of a television PSA about brown treesnakes and other snakes.

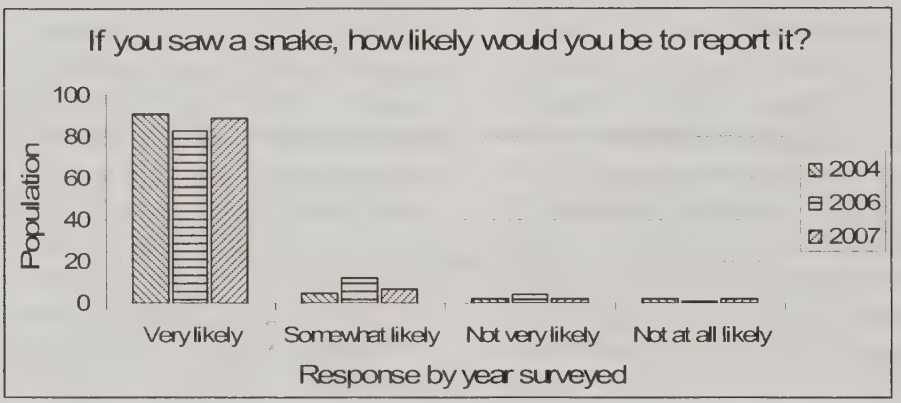



**Figure 3.** The intent to report a snake sighting remains high when total population responses (n=501) are averaged, yet if this were plotted by age, it would reveal that fewer 18-34 year olds would be likely to report a snake compared to older age groups.

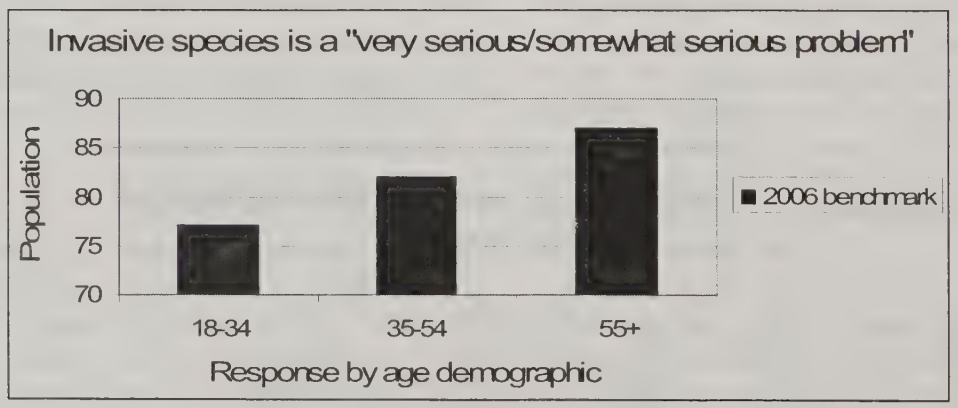



**Figure 4.** When plotted by age demographic, fewer 18-34 year olds view invasive species as serious problem, compared to older demographic groups.




**Figure 5.** Website page request records for 643-PEST.com show that there were 2421 unique webpage viewers within two months of this targeted outreach effort, with spikes stemming from outreach via networking activities popular with Generation Y.

brown treesnake. The video of the woman portraying the brown treesnake shows her in body make-up slithering around like a brown treesnake. The video is intercut with real brown treesnake video and photos, while a voiceover talks about the threat of these and other snakes pose to Hawaii.

To draw viewers to this website, two gregarious young women were hired to hand out "movie promotion" cards to their peers at local events, such as the import car show, hip hop events, and the Bob Marley Fest concert. In the early morning hours and for several days following these events, webpage requests would spike by as many as 649 unique visitors, for a total of 2421 unique visitors during the two month trial (Figure 5). MySpace profiles were also set up for each of the six women, with photos of them in character and links back to the website. MySpace "friends" totaled 374, and 1,054 people visited one or more of these MySpace pages to view their profiles.

## DISCUSSION

The results of the public awareness surveys reflect the relative success of concerted outreach efforts using short-term media campaigns, opportunistic news media stories, and other techniques. Survey results also highlight the challenges of educating the public and fostering desired actions, particularly when different messages require multiple delivery methods. Careful planning and testing of messages with focus groups can help ensure that messages are appropriate, however, there may be no way to ensure that outreach methods are appropriately matched with messages and audiences.

There may also be messages and methods that are not appropriate for widespread public outreach, such as the messages that need to be sent to the small subset of the population that continues to smuggle in and harbor snakes. Even with all of the money and effort spent on educating the public, there is another message that has been received and spread among this group—although there are penalties, it is relatively easy and safe to smuggle snakes into Hawaii (Kraus and Cravalho 2001). This issue must be addressed, or this unintended message will continue to spread amongst that portion of the population that does not view it as a serious issue. Furthermore, repeated snake sightings and news of snakes found roaming free or turned in to the amnesty program may have a negative effect on the currently supportive public.

Although the targeted outreach campaign to 18-34 year olds allowed CGAPS to experiment with unique methods for message delivery, it has not been determined whether the target audience is becoming more aware or concerned of the problem, or if the campaign will result in positive response actions. This is not to say that there is not merit to continuing this experiment in some form. CGAPS could wait to see if awareness and concern rises as this demographic group ages, but that would be a gamble.

Furthermore, the huge popularity of social networking and electronic media should not be ignored or thought of as a passing fad. Although

the 643-Pest.com website site may be considered too sexually explicit for widespread public outreach in its current form, efforts will be made to preserve its appeal to the targeted audience, while making it acceptable in the event that the site actually does reach a "tipping point."

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## LITERATURE CITED

GLADWELL, M. 2000. The tipping point: how little things can make a big difference. Little, Brown and Company. Time Warner Book Group, New York, New York, USA.

KRAUS, F., AND D. CRAVALHO. 2001. The risk to Hawaii from snakes. Pacific Science 4:409-417.

Q-MARK RESEARCH AND POLLING. 2007. Statewide invasive species study. Report prepared for CGAPS, February 2007. Q-Mark Research and Polling, Honolulu, Hawaii, USA.

STONE, C. P. 1985. Alien animals in Hawaii's native ecosystems: Toward controlling the adverse effects of introduced vertebrates. Pages 251-279 *in* P. P. Stone and J. M. Scott, editors. Hawaii's terrestrial ecosystems: preservation and management. University of Hawaii Press for University of Hawaii Cooperative National Park Resources Studies Unit, Honolulu, Hawaii, USA.

THE KITCHENS GROUP. 1996. Issue analysis: Hawaii statewide. Report for The Nature Conservancy of Hawaii. The Kitchens Group, Orlando, Florida, USA.

THE NATURE CONSERVANCY OF HAWAII, AND NATURAL RESOURCES DEFENSE COUNCIL. 1992. The alien pest species invasion in Hawaii: background study and recommendations for interagency planning. Joint Agency Report.

U.S. CONGRESS OFFICE OF TECHNOLOGY ASSESSMENT. 1993. Harmful non-indigenous species in the United States, OTA-F-565. Washington, D.C., USA. Government Printing Office, September 1993.

WARD RESEARCH. 2003. Focus groups among Oahu and neighbor island homeowners: summary of focus group findings. Report prepared for CGAPS. Ward Research, Inc., Honolulu, Hawaii, USA..

WARD RESEARCH. 2004. Baseline awareness of, and attitudes toward, invasive species in Hawaii: a telephone survey of residents statewide. Report prepared for CGAPS, August 2004. Ward Research, Inc., Honolulu, Hawaii, USA.

WARD RESEARCH. 2006. Tracking awareness of, and attitudes towards, invasive species in Hawaii: a telephone survey of residents statewide. Report prepared for CGAPS, June 2006. Ward Research, Inc., Honolulu, Hawaii, USA.

# HABITATTITUDE™: GETTING A BACKBONE ABOUT THE PET RELEASE PATHWAY

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***Abstract:*** Many people would not consider their family complete without a pet. Unfortunately, for some pet owners, circumstances arise that prevent them from being able to properly care for their companions and pet abandonment has become one of the most common pathways of vertebrate species introduction. It is also one of the most challenging pathways to address—once the animals become established, eradication and control programs face significant public scrutiny and are often challenged by "animal rights" groups. Prevention measures are thus the key to minimizing the size and impacts of the "pet release pathway." Habitattitude™ is a proactive campaign designed and implemented by the Pet Industry Joint Advisory Council and partners to prevent the introduction of unwanted pets into natural systems. The campaign has three components: (1) educating consumers to make wise pet choices, (2) providing resources to enable high standards in animal care and maintenance, and (3) encouraging pet owners to choose among several alternatives to the release of their pets if problems do arise. Habitattitude™ messages are being promoted in pet stores, product advertisements, industry trade shows, and industry-relevant magazines.

***Key Words:*** education, Habitattitude™, industry, invasive species, pathway, pets, public-private partnerships.

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## INTRODUCTION

Many people would not consider their family complete without a pet – whether it be a dog (*Canis lupus familiaris*), cat (*Felis silvestris catus*), parakeet (*Melopsittacus undulates*), goldfish (*Carassius auratus*), or bearded dragon (*Pogona vitticeps*). The American Pet Product Manufacturers Association (APPMA) estimates that there are 360 million pets in the United States (US) and nearly 63% of American households have at least one companion animal (APPMA 2007a). The same percentage holds for Australian households, with 53% owning a cat or dog (Hill 2006). In the United Kingdom, approximately half of the households include pets (McNicholas et al. 2005). Pet purchase and care supports a thriving industry, with the annual market value estimated at $40.8 billion in the US alone (APPMA 2007b).

Pets bring considerable joy and security into people's lives and studies show that their companionship substantially benefits human health and wellbeing (Barker 1999, Mayon-White 2005). The emotional bond between owner and pet can be as vital to the owner as many human relationships and confer similar psychological benefits (Voith 1985, McNicholas et al. 2005). A sampling of statistics reflecting the depth and scope of the human-animal bond is presented in Table 1.

Although research results are not always consistent (e.g., Parslow and Jorn 2003) and vary among species (Friedman 1995), health benefits commonly attributed to pet ownership include: stress reduction (Friedman et al. 1983, Katcher 1984), reduced risk of cardiovascular disease (Anderson et al. 1992, Patronek and Glickman 1993), higher survival rates from myocardial infarction (Friedman et al. 1980), reduced risk of asthma and allergic rhinitis in children exposed to pets during the first year of life (Nafsted et al. 2001, Ownby et al. 2002), and better physical and psychological well-being in older people (Siegel 1990, Raina et al. 1999). Research has also demonstrated significantly less sickness-related school absenteeism among children who live with companion animals (McNicholas et al. 2005). The psychiatric profession is increasingly employing pets for their therapeutic benefits (Barker 1999).

Pets have the potential to foster better people. Through pet ownership, children can learn to take responsibility, as well as extend care and love to others. Studies indicate pets may contribute to a child's sense of identity, autonomy and initiative, industriousness, and trust (Bryant 1990, Robin and Ten Bensel 1990, Brown et al. 1996). Children brought up with pets show better self-esteem, social skills, and empathy with others than children with

**Table 1.** Statistics reflecting the scope of the human-animal bond.

1. 99% of pet owners consider their pets family members (Voith 1985)
2. 83% refer to themselves as their pet's mom or dad (AAHA 2001)
3. 63% of pet owners say "I love you" to their pet at least daily (AAHA 2001)
4. 59% celebrate their pet's birthday (AAHA 2001)
5. Children are more likely to have pets than siblings or fathers (Melson 2001)
6. 57% would prefer their pet as their only companion if stranded on a desert island (APPMA 2001/2002)
7. 52% believe their pets listen to them best (AAHA 2001)

no pets (Poresky and Hendrix 1990, Van Houtte and Jarvis 1995).

Due to changing demographic patterns, pets are increasingly the only animals that people have an opportunity with which to interact (Katcher and Beck 1987, Louv 2005). Katcher and Beck (1988) argued that caring for pets and other animals gives rise experiences of nurturing and being nurtured that are increasingly lacking in interchanges among people. Sobel's (1996) developmental approach to ecological literacy indicates that 4-7 year-old children exposed to pets are more likely to build empathy for and a sense of connectedness to "creatures" (live and imaginary), suggesting that people who interact with pets may be more likely to develop an interest in wild animals and protecting the environment. Marks et al. (1994) found that scores associated with the level of pet owner attachment were significantly and positively correlated with generativity (concern for the next generation), further supporting the theory that pet owner's may be more likely to care about environmental issues. Unfortunately, for some pet owners, circumstances arise that prevent them from being able to properly care for their companions (Table 2). Some well-meaning pet lovers try to solve such problems by setting their pets free, i.e., releasing them into the natural environment (Courtenay 1999, Fuller et al. 1999). Of course, there are also individuals who abandon pets as a result of callousness or boredom. According to APPMA's 2007-2008 Pet Owner's Survey (APPMA 2007a), pet abandonment is the least common choice when pet owner's make arrangements for a pet for which they can no longer provide (Table 3).

For the pets, "freedom" is often a traumatic experience; they may not be able to find adequate food and shelter, and may become vulnerable to other animals, traffic, and people who consider them a nuisance. Abandoned pets that do survive can cause significant harm to the environment by preying on or competing with native fish and wildlife, spreading disease and parasites, and destroying fragile habitats (Moyle 1996, Genovesi and Bertolino 2001, Algar et al. 2002). In short, they can become invasive species: non-native species that cause harm, or have the potential to cause harm, to the environment, economies, or human health (Federal Register 1999).

Invasive species are one of the most significant drivers of environmental change worldwide (Sala et

**Table 2.** Reasons people give up their pets.

1. A family member develops allergies
2. The owner's lifestyle changes unexpectedly
3. Housing location and/or policies change
4. The pet outgrows its housing
5. The animal's behavior becomes problematic
6. The pets reproduce and are too many to care for
7. The animal becomes sickly (and costly)
8. The pet's needs are not compatible with the owner's wants
9. Children leave home or develop other interests
10. Fear of zoonotic disease transmission

**Table 3.** Alternative arrangements identified, by percentage, by pet owners when they are unable to provide continual animal care.

| | Pet Choice and No. Surveyed | | | | |
|---|---|---|---|---|---|
| | Dog | Cat | Bird | Small Animal | Reptile |
| Selected | 580 | 492 | 342 | 301 | 333 |
| Arrangements | % | % | % | % | % |
| Give to friend/relative | 74 | 71 | 77 | 68 | 61 |
| Bring to humane society | 12 | 15 | 8 | 15 | 10 |
| Give to school | 1 | * | 6 | 12 | 19 |
| Other (e.g., sell) | 4 | 4 | 8 | 1 | 14 |
| Bring to shelter | 7 | 9 | 4 | 4 | 4 |
| Bring to vet's office | 3 | 5 | 2 | 3 | 2 |
| Put to sleep | 5 | 4 | 2 | 1 | - |
| Abandon | * | - | * | 6 | 1 |

*Less than 0.5%. Adapted from APPMA 2007a.

al. 2000, McNeely et al. 2001). They have been implicated in the endangerment of specific species (Wilcover et al. 1998), degradation of aquatic and terrestrial environments (Cartlon 2001, D'Antonio and Kark 2002), and the alteration of biogeochemical cycles (D'Antonio and Vitousek 1992, Mack and D'Antonio 1998). Consequently, they can contribute to social instability and economic hardship, placing constraints on the conservation of biodiversity, sustainable development, and economic growth (McNeely 2001, Pimentel 2002, GISP 2006). The globalization of trade, travel, and transport is greatly increasing the number of invasive species (both individual animals and species) that are being moved around the world, as well as the rate at which they are moving (McNeely et al. 2001, Burgiel et al. 2006). At the same time, changes in climate and land use are rendering some habitats more susceptible to biological invasion (Mooney and Hobbs 2000).

Pet abandonment has become one of the most common pathways of vertebrate species introduction (Courtenay 1999, Fuller 2003, Kraus 2003) and one of the most challenging to address once the animals become established. Eradication and control programs face significant public scrutiny and are often challenged by "animal rights" groups. In some cases, natural resource managers have been unable to eradicate or control these feral invasives due to interference and threats presented by "animal rights" advocates (e.g., Genovesi and Bertolino 2001). Prevention measures are thus the key to minimizing the size and impacts of the "pet release pathway." To be successful, these initiatives need to target the riskiest aspect of the pathway, namely pet owners, and carefully consider the motivators and implications of the human-animal bond.

## THE CAMPAIGN

Public education is a major tool to minimize pet releases (Wittenberg and Cock 2001). In 2005, The Pet Industry Joint Advisory Council (PIJAC) joined with the US Fish and Wildlife Service (USFWS) and the National Oceanic and Atmospheric Administration (NOAA) Sea Grant program to create Habitattitude™, a proactive campaign designed to prevent the release of unwanted pets. This innovative, partnership-based initiative emerged out of the shared vision and commitment of several US federal and state agencies, as well as the pet and aquaria industries. The program initially focused on aquatic species and promoted a "Protect our environment: Do not release fish and aquatic plants" message (www.habitattitude.net/). Habitattitude™ is now growing to address all pets, engage an even wider range of additional affiliates (over 70 as of July 2007), and expand its messaging. To have a Habitattitude™ is to "Do right by your pet. Do right by our environment."

Both PIJAC and its Canadian counterpart have launched consumer-focused websites (www.pijac.org/habitattitude and

www.habitattitude.ca/en/aboutus/, respectively) that promote and provide information on the three primary goals of the campaign: (1) protect the natural environment (habitats) from the impacts of unwanted pets (i.e., potentially invasive species), (2) ensure that pets are carefully selected and well cared for (habits), and (3) help pet lovers find alternatives to the release of their pets (attitudes).

The first goal is supported by messaging that addresses the later two. In brief, the campaign communicates the following information to consumers.

**Before Selecting a Pet**

When it comes to pets, a quick decision is often a poor decision. Ideally, a pet lover carefully considers how to be a responsible pet owner and environmental steward (caretaker) before choosing an animal companion. A person's intent for a companion, lifestyle, family composition, work schedule, income, health and physical fitness, and even their personality will make some pets more suitable for them than others.

**Pet Care**

Healthy pets make for happy pet owners. When a person brings a pet into their home, they become its care taker – it is their responsibility to provide for its every need – proper diet, clean water, adequate housing and shelter, exercise, entertainment, and even emotional well-being. While many companion animals are short-lived, others can live for decades and some (turtles and parrots, for example) can live for more than 100 years. And, although most pets are small and remain small, others will grow to very large sizes and require specialized feeding and housing. The amount of joy a person receives from having a pet and the pet's well-being are completely up to the person. Care for the pet properly and both person and pet will benefit from a healthy relationship.

**Responsible Pet Placement**

Releasing a pet into the natural environment puts both the pet and the environment at risk. Furthermore, in many locations, it is illegal to release a pet into the natural environment. Unfortunately, for some pet lovers, circumstances do arise that prevent them from being able to properly care for their pet companions. (Table 2). A number of alternatives to the release of unwanted pets are available in these cases. See Table 4 for pet placement guidelines.

**Table 4.** Pet placement guidelines

| |
|---|
| 1. Give/trade to a friend, relative, another pet owner or hobbyist. |
| 2. Bring to humane society or animal shelter. |
| 3. Contact pet retailer for advice or possible returns. |
| 4. Donate to a hobby group, school, or relevant business. |
| 5. Contact veterinarian for guideline on humane euthanasia. |
| 6. Seal all associated aquarium/terrarium plants in plastic bags and dispose of them in the trash. |

Note: These guidelines are adapted from the "Alternatives to Release" adopted by the Aquatic Nuisance Species Task Force for fish and plants: www.habitattitude.net.

Through the PIJAC websites, supportive information tools (e.g., articles, guidelines, and website links) are provided. Consumers are also encouraged to discuss their pet options, care, and placement needs with local veterinarians, animal breeders, pet store staff, hobbyists, and friends who have pets, as well as to consult relevant books and magazines.

## SPECIAL FOCUS PROGRAMS

The campaign has developed a special focus on two aspects of consumer activities that pose the greatest risks for pet release.

**Water Gardening Hobby**

Backyard pond and water gardening is the fastest growing segment of both the pet and horticulture industries and involves the intentional release of animals (most commonly fish) into outdoor environments. PIJAC is working with the American Nursery and Landscape Association to create codes of conduct for water gardening retailers and consumers.

At least 5% percent of the US population is estimated to have a water garden, for a total of more than 15 million water gardens in the US alone (Fins and Flowers 2007). Water gardens add beauty and tranquility to backyards and other landscapes. However, if they are not established or maintained thoughtfully, water gardens can also become a source of environmental problems. Despite best intentions, many water gardeners unknowingly introduce harmful animals, plants, and diseases to

their ponds. When these introduced organisms (especially fish and small, floating plants) find their way into natural waterways, they can cause substantial impacts to native species and their habitats. The following section provides messages that can be shared with water gardeners.

In designing water gardens, carefully consider aesthetic desires, as well as the local environmental conditions (seasonal rainfall, for example), neighborhood setting, and budget. Refrain from establishing the water garden within, connected to, or in the close vicinity of natural water bodies (ponds and streams, for example). This will help prevent the unintended introduction of harmful plants and animals into the natural environment. Ensure that the pond pets and plants are well-contained and thus can not get washed away during rainstorms or flood events into local water bodies or public sewer systems (which often drain into local water bodies).

Consider the likelihood that children will collect plants and animals in the pond and release them elsewhere. As is necessary, plan for a fence, relevant signs, and to say "no" to requests to collect from the water garden.

It is important to carefully evaluate the costs and budget. Many people are surprised at the cost of creating and maintaining their water gardens – pond liners, pumps, stones, lights, plants, and fish can be quite expensive (some popular fish can cost more than US$100 each). Pond maintenance requires both additional money and time. A prospective water gardener needs to be sure that they can properly care for the pond pets and plants as long as they own the property.

Water gardening is both an art and a science. When it comes to water garden pets and plants, a quick decision is often a poor decision. When ready to select pond pets and plants, be aware that many horticulture and pet product magazines and websites sell or promote plants and animals that are known to be invasive, or have the potential to become invasive. In some states, it is illegal for these species to be bought and sold. Contact state native plant societies, cooperative extensive agencies, or fish and game departments to find out what plants and animals are the safest for water gardens in the area.

Learn the biology and needs of the plants and animals being considered for the water garden. What care do they require? How large do they grow? Are they likely to reproduce? Are they likely to escape or be moved from the pond (by wild animals or neighborhood children)? What diseases are they prone to and how are they treated? What environmentally-friendly options exist if the plants, fish, and other animals get too big or too numerous?

If a water gardener is faced with needing to find a new home for a pond pet, they can choose among the options in Table 4. Releasing pond pets into the natural environment puts these fish and other animals, as well as the natural environment at risk. Furthermore, in many locations, it is illegal to release water garden pets and plants into the natural environment.

**Classroom Activities**

Pets are often maintained in classroom settings, especially in elementary schools. They fascinate and entertain, and can become "teachers" themselves – on topics ranging from art to science to personal hygiene. Having a pet in the classroom is, however, a significant responsibility for teachers; pets in the classroom need time, attention, and financial investments that are often above and beyond required duties and budget. It is up to the teacher to ensure that the animals receive the necessary care, and that they create a safe and enjoyable environment for both the students and the pets. The first step in responsible classroom pet care is to consider what kind of pet is most appropriate before bringing an animal into the school environment – the animal's particular needs and behaviors, the age of the students, school schedules (including long holiday breaks), and the teacher's available time and budget are all important variables to consider. The teacher can even make the selection process a learning exercise for the students.

Teachers, perhaps even more often than parents, find themselves faced with the need to locate a new home for a pet. What to do with the animal over summer break? What happens if a child develops allergies, is fearful of animals, or perhaps gets injured? What if school policies on classroom pets change? What if the classroom pets produce too many offspring or grow too large and costly? Who will adopt the chicks hatched out at Easter or the tadpoles raised from eggs?

Questions such as these are ideally answered before choosing a classroom pet. However, surprises do happen and sometimes well-meaning teachers will attempt to do the right thing by releasing the pet into the natural environment. This is neither the best solution for the pet, nor the environment. A person is faced with needing to find a new home for a classroom pet please refer to

Table 4 in order to make the best possible decision for the animal, and make it a learning experience for the students as well.

## IMPLEMENTATION

PIJAC's focus is primarily to implement the Habitattitude™ campaign through the pet industry and pet owning public in the US and Canada, and secondarily to join with program partners and associates in extending the campaign into other sectors and countries. In its first two years of operation, the campaign has made numerous accomplishments (Table 5).

These serve as the foundation for new and upcoming campaign initiatives, including: (1) additional focus on terrestrial species (including new branding and partnership building); (2) supplemental fact sheets, booklets, coloring books and other materials available through the PIJAC (US) website; (3) promotion of the Habitattitude™ brand and message through a growing number of retail store venues and pet products (e.g., store signage, door decals, fish bags, and pet starter kits); (4) inclusion of Habitattitude™ message in PIJAC best practice manuals for store employees (e.g., Doing it Right: A Pet Store Employee's Guide to Professional Success [and Fun!]) and hobbyists; (5) targeted, site-specific partnerships (e.g., with the State of Florida and National Park Service in the Everglades); and, (6) expanded partnerships with the media and celebrity spokespersons. PIJAC offers the following lessons learned to individuals who would like to help promote the campaign messages or initiate campaigns in other countries: (1) focus on communicating the key messages (as described in this paper) to pet owners; (2)

**Table 5.** Examples of Habitattitude™ accomplishments, 2005-2007.

1. Enthusiastic endorsement and support from the PIJAC Board of Directors and other pet industry trade associations, which includes high-level industry representatives from major manufacturers, distributors, and retailers in the pet industry (e.g., PetSmart, PETCO, Central Garden and Pet Supply, Marineland).

2. Enthusiastic endorsement and support from key federal government agencies working on invasive species issues, including the Department of the Interior, Department of Commerce, and Department of State. In the Department of Interior's 2005 report to Congress, the Department explicitly sited engagement in Habitattitude™ as one of its most significant contributions to environmental conservation.

3. Received start up funds from USFWS to establish the Habitattitude™ brand, brand marketing campaign, and brand impact monitoring.

4. USFWS, PIJAC, and NOAA Sea Grant Program were awarded a two-year seed grant for engaging academic and state agencies in the program. Although monetarily small, the seed grant was itself a significant accomplishment for the campaign: it marked the first time that the pet industry had engaged with federal and state government in such a proactive, large-scale public education and outreach program.

5. From 2004-2007, program partners presented Habitattitude™ poster displays at pet industry trade shows, consumer pet shows, numerous aquarium society meetings, the Outdoor Writers Association annual meetings, meetings of all 50 state fish and game agencies, and a number of environment meetings dealing with invasive species issues.

6. Exceptional support from campaign affiliates. For example, PETCO promotes the campaign in newspaper inserts that reach 30-34 million households/month, giving away additional copies of the inserts at their 850+ stores, and placing Habitattitude™ signage, care sheets, and other promotional materials in all their stores. They are now planning to distribute campaign materials in every aquarium they distribute as "PETCO private label starter kits."

7. Adoption and extension of the Habitattitude™ campaign by PIJAC Canada (www.habitattitude.ca/en/aboutus/)

communicate the message in such as way as to move people toward solutions (choosing, caring, and rehoming pets wisely) as a means of moving them away from problems (pet release) (i.e. it's not enough to say "Don't Release," provide clear directives for the appropriate behavior); (3) work through or closely with pet industry trade associations, corporate leaders, and magazine publishers; (4) when working with the pet industry, be business-oriented in the approach (e.g., use strategic planning and market-based incentives models, and invest in brand research, design, and marketing); (5) recognize that the pet industry is diverse (from large corporations to small "Mom and Pop" stores) and that there is considerable diversity in business approach (e.g., some stores invest considerable amounts of money in signage while others believe it ineffectual). Thus, be sure to build flexibility into the campaign implementation strategy; (6) employ an overall approach based in social marketing and communication psychology (i.e. learn to understand the audience and what motivates them); (7) become familiar with pet ownership surveys and human-animal bond studies so as to understand the connections between pets and people in the area; (8) work with natural resources managers and survey the scientific literature to identify types of pets and locales that are high risk for release; (9) make a long-term commitment to campaign implementation and be patient with potential partners who might initially be defensive or not understand the significance of the issue or their role in relation to it; and, (10) try to make it fun for all involved.

## CONCLUSION

As is true of all human activities, the benefits of pet ownership are not without risks and impacts. The degree to which pet ownership confers benefits to human society is a reflection of pet owners' attendance to individual responsibilities, and the pet industry recognizes that Habitattitude™ alone will not fully address the risks associated with the "pet release pathway." Science-based regulation may be necessary to prevent potential impacts by specific high-risk species. The pet industry has, for example, recently supported a ban on the importation of the Gambian giant pouch rat (*Cricetomys gambianus*) and regulation (including permitting and microchipping) of large constrictors (*Boa* spp.) due to invasion concerns. PIJAC has also encouraged the development of screening processes for first time introductions of all species, irregardless of their intended use. The pet industry maintains its interest in working with regulators and other stakeholders on the development of science-based policies and regulations and is currently engaged in processes associated with the National Invasive Species Council, Invasive Species Advisory Committee, Aquatic Nuisance Species Task Force and its associated regional panels, the Commission for Environmental Cooperation, and state initiatives, such as a Florida Conservation Commission-led risk analysis of marine ornamental fish. PIJAC would welcome the opportunity to work with similar bodies in other countries.

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## LITERATURE CITED

AAHA (American Animal Hospital Association). 2001. 2001 AAHA Pet Owners Survey. American Animal Hospital Association, Lakewood, Colorado, USA.

Algar, D., A. Burbidge, and G. Angus. 2002. Cat eradication on Hermite Island, Motebello Islands, Western Australia. Pages 14-18 *in* C. Veith and M. Clout, editors. Turning the tide: The eradication of invasive species. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland.

Anderson, W. P., C. M. Reid, and G. L. Jennings. 1992. Pet ownership and risk factors for cardiovascular disease. Medical Journal of Australia 157:298-301.

APPMA (American Pet Product Manufacturer's Association). 2001. 2001-2002 APPMA national pet owners survey. American Pet Product Manufacturer's Association, Greenwich, Connecticut, USA.

APPMA (American Pet Product Manufacturer's Association). 2007a. 2007-2008 APPMA national pet owners survey. American Pet Product Manufacturer's Association, Greenwich, Connecticut, USA.

APPMA (American Pet Product Manufacturer's Association). 2007b. APPMA unpublished data. American Pet Product Manufacturer's Association, Greenwich, Connecticut, USA.

Barker, S. 1999. Therapeutic aspects of the human-companion animal interaction. Psychiatric Times XVI, Issue 2. http://www.psychiatrictimes.com/p990243.html).

Brown, B., H. Richards, and C. Wilson. 1996. Pet bonding and pet bereavement among adolescents. Journal of Counseling and Development 74:505-509.

Bryant, B. 1990. The richness of the child-pet relationship: a consideration of both benets and costs of pets to children. Antrhozoýs 3:253-261.

Burgiel, S., G. Foote, M. Orellana, and A. Perrault. 2006. Invasive alien species and trade: integrating prevention measures and international trade rules. Center for International Environmental Law and Defenders of Wildlife, Washington, D.C., USA

Carlton, J. 2001. Introduced species in U.S. coastal waters: environmental impacts and management priorities. Pew Oceans Commission, Alexandria, Virginia, USA.

Courtenay, JR., W. 1999. Aquariums and water gardens as vectors of introduction. Pages 127-128 *in* R. Claudi and J. Leach, editors. Nonindigenous freshwater organisms: vectors, biology and impacts. Lewis Publishers, Boca Raton, Florida, USA.

D'Antonio, C. and S. Kark. 2002. Impacts and extent of biotic invasions in terrestrial ecosystems. Trends in Ecology & Evolution 17:202-204.

D'Antonio, C. and P. Vitousek. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and Systematics 23:63-87.

Federal Register. 1999. Executive Order 13112: Invasive Species. Office of the President. Federal Register 64:6183-6186.

Fins and Flowers Water Garden LLC. 2007 (24 July 2007; http://www.finsandflowers.com/index.php?option=com_content&task=view&id=8&Itemid=9).

Friedman, E. 1985. The role of pets in enhancing human well-being: physiological effects. Pages 33-53 *in* I. Robinson, editor. The Waltham book of human-animal interaction: benefits and responsibilities of pet ownership. Elsevier Science Inc., New York, USA.

Friedman, E., A. Katcher, J. Lynch, and S. Thomas. 1980. Animal companions and one year survival of patients after discharge from a coronary care unit. Public Health Report 95:307-312.

Friedman, E., A. Katcher, J. Lynch, and S. Thomas. and P. Messent. 1983. Social interaction and blood pressure: the influence of animal companions. Journal of Nervous and Mental Disease 171:461-465.

Fuller, P. 2003. Freshwater aquatic vertebrate introductions in the United States: patterns and pathways. Pages 123-151 *in* G. Ruiz and J. Carlton, editors. Invasive species: vectors and management strategies. Island Press, Washington, D.C., USA.

Fuller, P., L. Nico, and J. Williams. 1999. Non-indigenous fishes introduced into inland waters of the United States. American Fisheries Society, Special Publication 27, Bethesda, Maryland, USA.

Genovesi, P. and S. Bertolino. 2001. Human dimension aspects in invasive alien species issues: the case of the failure of the grey squirrel eradication project in Italy. Pages 113-120 *in* J. McNeely, editor. 2001. The great reshuffling: human dimensions of invasive alien species. The World Conservation Union-IUCN, Gland, Switzerland:

GISP (Global Invasive Species Programme). 2006. Invasive species and poverty: exploring the links. The Global Invasive Species Programme, Capetown, South Africa.

Hill, M. 2006. Contribution of the pet care industry to the Australian economy. BIS Shrapel Pty Ltd., Sydney, Australia.

Katcher, A. and A. Beck. 1987. Health and caring for living things. Antrhozoýs 1:175-183.

Katcher, A. and A. Beck. 1988. Health and caring for living things. *In* A. Rowan, editor. Animals and people sharing the world. University Press of New England, Hanover, Massachusetts, USA.

Katcher, A., A. Beck and H. Segal. 1984. Contemplation of aquarium for the reduction of anxiety. Pages 171-178 *in* R. Anderson, B. Hart, and L. Hart, editors. The pet connections: its influence on our health and quality of life. Global Publishing Company, St. Paul, Minnesota, USA.

Kraus, F. 2003. Invasion pathways for terrestrial vertebrates. Pages 68-92 *in* G. Ruiz and J. Carlton, editors. Invasive species: vectors and management strategies. Island Press, Washington, D.C., USA

Louv, R. 2005. Last child in the woods: saving our children from nature-deficit disorder. Algonquin Books of Chapel Hill, Chapel Hill, North Carolina, USA.

Mack, M. and C. D'Antonio. 1998. Impacts of biological invasions on disturbance regimes. Trends in Ecology and Evolution 13:195-198.

Marks, S., J. Koepke, and C. Bradley. 1994. Pet attachment and generativity among young adults. The Journal of Psychology 128:641-651.

Mayon-White, R. 2005. Commentary: pets-pleasures and problems. British Medical Journal 331:1254-1255.

McNeely, J. 2001. The great reshuffling: human dimensions of invasive alien species. The World Conservation Union-IUCN, Gland, Switzerland.

McNeely, J., H. Mooney, L. Neville, P. Schei, and J. Waage. 2001. Global strategy on invasive alien species. The World Conservation Union-IUCN in collaboration with the Global Invasive Species Programme, Cambridge, United Kingdom.

McNicholas, J., A. Gilbey, A. Rennie, S. Ahmedzai, J. Dono, and E. Ormerod. 2005. Pet ownership and human health: a brief review of evidence and issues. British Medical Journal 331:1252-4.

Melson, G. 2001. Why the wild things are: animals in the lives of children. Harvard University Press, Cambridge, Massachusetts, USA.

Mooney, H. and R. Hobbs. 2000. Invasive species in a changing world. Island Press, Washington, D.C., USA.

MOYLE, P.B. 1996. Effects of invading species on freshwater and marine ecosystems. Pages 86-92 *in* O. Soutland, P. Shea, and A. Viken, editors. Proceedings of the Norway/UN Conference on Alien Species, Trondheim, 1-5 July, 1996. Directorate for Nature Management and Norwegian Institute for Nature Research, Trondheim.

NAFSTED, P., P. MAGNUS, P. GAADER, AND J. JAAKOLA. 2001. Exposure to pets and atopy-related disease in the first 1 years of life. Allergy 157:298-301.

OWNBY, D., C. JOHNSON, AND E. PETERSON. 2002. Exposure to dogs and cats in the first year of life and risk of allergic sensitizations at 6 to 7 years of age. Journal of the American Medical Association 288:963-972.

PARSLOW, R. AND A. JORN. 2003. Pet ownership and risk factors for cardiovascular disease: another look. Medical Journal of Australia 179:166-168.

PATRONEK, G. AND L. GLICKMAN. 1993. Pet ownership protects against the risks and consequences of coronary heart disease. Medical Hypotheses 40:245-249.

PIMENTEL, D. 2002. Biological invasions: economic and environmental costs of alien plant, animal, and microbe species. CRC Press, Washington, D.C., USA.

PORESKY, R. AND C. HENDRIX. 1990. Differential effects of pet presence and pet-bonding on young children. Psychology Report 67:51-54.

RAINA, P, D. WALTER-TOEWS, B. BONNET, C. WOODWARD, AND T. ABERNATHY. 1999. Influence of companion animals on the physical and psychological health of older people: an analysis of a one-year longitudinal study. Journal of the American Geriatric Society 17:323-329.

ROBIN, M. AND R. TEN BENSEL. 1990. Pets and the socializations of children. Lathum Letter 11:1-23.

SALA, O., F. CHAPIN III, J. ARMESTO, E. BERLOW, J. BLOOMFIELD, R. DIRZO, E. HUBER-SANWALD, L. HUENNEKE, R. JACKSON, A. KINZIG, R. LEEMANS, D. LODGE, H. MOONEY, M. OESTERHELD, N. POFF, M. SKYES, B. WALKER, M. WALKER, AND D. WALL. 2000. Global biodiversity scenarios for the year 2100. Science 287:1770-1774.

SIEGAL, J. 1990. Stressful life events and use of physician services among the elderly: the moderating role of pet ownership. Journal of Personality and Social Psychology 58:1081-1086.

SOBEL, D. 1996. Beyond ecophobia: reclaiming the heart in nature education. Great The Orion Society and the Myrin Institute, Barringon, Massachusetts, USA.

VAN HOUTTE, B. AND P. JARVIS. 1995. The role of pets in preadolescent psychosocial development. Journal of Applied Developmental Psychology 16:463-479.

VOITH, V. 1985. Attachment of people to companion animals. Veterinary clinics of North America: small animal practice 15:289-295.

WILCOVER, D., D. ROTHSTIEN, J. DUBOW, A. PHILLIPS AND E. LOSOS. 1998. Quantifying threats to imperiled species in the United States. BioScience 48:607-615.

WITTENBERG, R., AND M. COCK. 2001. Invasive alien species: a toolkit of best prevention and management practices. CAB International, Wallingford, Oxon, United Kingdom.

# CONSIDERING CONTROL OF INVASIVE BARRED OWLS TO BENEFIT CALIFORNIA SPOTTED OWLS: POSSIBLE JUSTIFICATION AND DRAFT METHODS

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***Abstract:*** Barred owls (*Strix varia*) recently expanded their range from eastern to western North America and are negatively affecting site occupancy, reproduction, and survival of federally threatened northern spotted owls (*Strix occidentalis caurina*), thereby confounding the land-based conservation strategy for northern spotted owls. Barred owls are moving southward into the range of non-federally listed California spotted owls (*S. o. occidentalis*), but are not, at this time or in the foreseeable future, a threat to the California spotted owl population. We recommend consideration of barred owl control in the northern Sierra Nevada Mountains while barred owl numbers are low enough to efficiently eliminate the possibility that they negatively affect California spotted owls outside of the foreseeable future.

***Key Words:*** barred owl, invasive species, spotted owl, *Strix occidentalis, Strix varia.*

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## INTRODUCTION

The California spotted owl (CSO, *Strix occidentalis occidentalis*) is the only subspecies of spotted owl that is not federally listed as threatened or endangered. CSOs are found in the Sierra Nevada Mountains of California and the mountains of southern California. Their range is situated geographically between that of the northern spotted owl (NSO, *S. o. caurina*), which ranges from British Columbia to northwestern California, and that of the Mexican spotted owl (*S. o. lucida*), which ranges from southern Utah and Colorado through Arizona, New Mexico, and Mexico. The NSO was listed as federally endangered in Canada in 1986 (Campbell and Campbell 1984, Government of Canada 2002) and as federally threatened in the United States under the Endangered Species Act (16 USC 1531 et seq.) in 1990 (USFWS 1990). The Mexican spotted owl was listed as federally threatened in the United States in 1993 (USFWS 1993). The CSO was identified as a "sensitive species" by the U.S. Forest Service (USFS) in the early 1970s (Beck and Gould 1992) and as a "species of special concern" by the State of California in 1978 (CDFG 1978).

Barred Owls (Strix varia) expanded their range from eastern to western North America, are negatively affecting NSOs (as described below), and are moving into the range of CSOs. Here we suggest consideration of barred owl control in the northern Sierra Nevada Mountains and methods to do so before barred owls increase in number, expand their range, and negatively affect CSOs.

## CONSERVATION OF THE NSO

Successful litigation prior to listing the NSO as a threatened species, protections under the Endangered Species Act after listing, and designation of approximately 2.8 million ha of critical habitat in western Washington, western Oregon, and northwestern California (USFWS 1992) significantly decreased timber harvest on federal lands within the range of the NSO during the early 1990s (Charnley 2006, Thomas et al. 2006). Interagency groups of biologists (Thomas et al. 2006) produced conservation strategies for the NSO (Thomas et al. 1990, Johnson et al. 1991, Thomas et al. 1993) during the early 1990s that culminated in the Northwest Forest Plan in 1994 (USDA and USDI 1994) in an attempt to conserve NSOs and other late-successional species while permitting some timber harvest on federal land. Of the 9.9 million ha of federally owned lands managed under the Northwest Forest Plan, 30.4% (3.01 million ha) was designated as Late-Successional Reserves to protect forests used by

NSOs, 10.7% (1.06 million ha) was designated as riparian reserves in large part to allow dispersal of NSOs among Late-Successional Reserves, 6.0% (0.60 million ha) was withdrawn for administrative purposes, and 23% (2.27 million ha) was designated as Matrix and Adaptive Management Area to remain available primarily for timber harvest (USDA and USDI 1994, Marcot and Thomas 1997). Harvests were permitted in younger forests in Late-Successional Reserves and riparian reserves when they were designed to enhance attainment of old-forest characteristics. A total of 29.9% (2.96 million ha) of the federal land included in the Northwest Forest Plan remained as National Parks or wilderness areas. Overall, "the federal forests in the Pacific Northwest underwent the largest shift in management focus since their creation" (Thomas et al. 2006)—from timber production to protection of late-successional forests used by NSOs and other species. As the basis of the Northwest Forest Plan, "no species in the United States has had a greater impact on land-use planning at the landscape scale" than the NSO (Noon and Blakesley 2006).

## INVASION BY BARRED OWLS

An invasive species, the barred owl, recently confounded this land-based conservation strategy. Historically, barred owls were distributed only in North America east of the Great Plains (Mazur and James 2000). Barred owls began their range expansion westward beginning more than 100 years ago according to early reports (K. Livezey and T. Fleming, unpublished data). The range expansion apparently was facilitated in large part by increased distribution of trees in the northern Great Plains (K. Livezey, unpublished data). Barred owls are now found in southern Canada from the Great Plains westward, in the Rocky Mountains of Idaho and Montana, and along the Pacific coast from southeastern Alaska to central California. They were first documented in British Columbia in 1943 (Rand 1944), Washington in 1965 (Rogers 1966), Oregon in 1974 (Taylor and Forsman 1976), and California in 1976 (B. Marcot, personal communication). The range of the barred owl now overlaps the entire range of NSOs in British Columbia (Hobbs 2005), Washington (Buchanan 2005), Oregon (Kelly and Forsman 2003), and northwestern California (see below).

Barred owls increased in number quickly throughout the range of the NSO. Two examples have been well documented. In Cowlitz Valley Ranger District of Gifford Pinchot National Forest in Washington, numbers of barred owl detections increased 8.6% annually from 1982–2000 (Pearson and Livezey 2003); more recently, the 98 barred owl sites known in 2001 increased to 143 occupied sites by 2006, thereby outnumbering the 111 sites known to be occupied by NSOs by 29% (Pearson and Livezey 2007). In Redwoods National and State Parks in California (K. Schmidt, personal communication), there were 36 NSO sites and 15 barred owl sites in 1995. By 2006, the situation essentially was reversed: there were 17 occupied NSO sites and 36 barred owl sites. We are not aware of an estimate of total numbers of barred owls in the Pacific Northwest. However, barred owls apparently outnumber NSOs within the specific range of the NSO in British Columbia and Washington and they greatly outnumber NSOs throughout western Washington, western Oregon, and northern California due to their large numbers in many locations where there are no NSOs (e.g., the Puget Sound area) or very few NSOs (e.g., southwestern Washington).

Barred owls negatively affect NSOs in several ways. Barred owls occasionally hybridize with spotted owls (Hamer et al. 1994, Haig et al. 2004, Kelly and Forsman 2004, Seamans et al. 2005), but this behavior is considered to be an "inconsequential" phenomenon that takes place mostly when barred owls move into new areas, and declines as barred owls become more numerous and have more access to other barred owls (Kelly and Forsman 2004). Barred owls are larger than NSOs, are physically aggressive toward them (E. Forsman, J. Mowdy, T. Snetsinger, G. Stagner, personal communication), apparently can kill them (Leskiw and Gutiérrez 1998), and use the same habitats and prey as they do (e.g., Hamer et al. 2001, Gremel 2005, Livezey 2007). Correlational analyses strongly suggest that barred owls negatively affect calling behavior (Olson et al. 2005, Crozier et al. 2006), site occupancy (Kelly et al. 2003, Pearson and Livezey 2003, Gremel 2005, Olson et al. 2005, Pearson and Livezey 2007), fecundity (Olson et al. 2004), and survival (Anthony et al. 2006) of NSOs.

The future of NSOs in the United States may be similar to what took place in Canada. The population of NSOs in Canada decreased from an estimated 500 pairs before European settlement to only 23 adult owls in 2005 due to habitat loss, habitat fragmentation, and competition with barred owls (Hausleitner and Blackburn 2006). Concerning NSOs in the United States, Gutiérrez et

al. (2004) placed nine possible outcomes of the threat from barred owls into three categories: clearly plausible, plausible, and not plausible or not clear. Their three clearly plausible outcomes were as follows: (1) barred owls will replace NSOs throughout the range of the NSO; (2) barred owls will replace NSOs in the northern (more mesic) part of their range; and (3) barred owls and NSOs will reach a competitive equilibrium favoring barred owls in most of the range of the NSO. The extinction of NSOs in the wild is the first outcome, and it is unknown whether a threatened species could sustain the population declines as described in the second and third clearly plausible outcomes.

The 2007 Draft NSO Recovery Plan (USFWS 2007) identified barred owls as a very important threat to NSOs. The Draft Plan listed many recovery actions relative to barred owls, including establish an Interagency BDOW Working Group, analyze existing data sets for effects of barred owls on NSOs, write a barred owl Management Plan, prioritize areas of control of barred owls, and conduct barred owl control experiments.

## CSO POPULATIONS AND THREATS

There are two distinct populations of CSOs. In the Sierras, the population of 1,865 cumulative territories of CSOs is largely contiguous (USFWS 2006) and appears to be stationary (Blakesley et al. 2006). Cumulative totals of all territories known to be historically or currently occupied by at least one CSO are used to indicate relative numbers, even though many of those territories probably are not currently occupied. In southern California, the 440–578 cumulative territories of CSOs occupy "islands" of high-elevation forests separated by lowlands of chaparral, desert scrub, and human development (Noon and McKelvey 1992, LaHaye et al. 1994). These islands comprise 15–20 populations with 3–270 individuals per population, and are separated from each other by 10–72 km (Verner et al. 1992, Gutiérrez 1994, LaHaye et al. 1994). These populations appear to be isolated from one another. No inter-mountain movements were documented for any of the 478 juvenile CSOs banded in the largest subpopulation of CSOs, which is in the San Bernardino Mountains (LaHaye et al. 2001). There is no complete, up-to-date analysis of population trends of CSOs anywhere in southern California. Most data for CSOs in southern California are from the San Bernardino Mountains (Franklin et al. 2004). Consistent surveying in that area was conducted only through 1998, and there are no consistent monitoring data for any of the other subpopulations of CSOs in southern California.

The U.S. Fish and Wildlife Service (USFWS) was petitioned in 2000 (CBD 2000) and 2004 (CBD 2004) to list the CSO as a threatened or endangered species. In both 12-month findings, USFWS (2003, 2006) determined that the species did not warrant listing. In the more-recent finding, the USFWS showed that (1) CSO populations were stationary in the Sierras and, as far as is known, in southern California; (2) regulatory mechanisms and protections were adequate; (3) catastrophic wildfire posed the greatest threat to CSO habitat but the treatment of fire fuels on USFS lands in the Sierra Nevada Mountains by the Sierra Nevada Forest Plan Amendment (USDA 2004) is addressing that threat; and (4) barred owls are not a current threat. Specifically, USFWS (2006) stated: "Although barred owls may pose a substantive threat to California spotted owls at some point in time, they do not appear to pose a significant threat now or in the foreseeable future."

For unknown reasons, barred owls have increased in numbers and distribution in the Sierras at a much slower pace than they did in northwestern California. As of April 2005, 496 barred owl detections had been recorded throughout California, only 60 of which (12%) were in the range of the CSO (Leskiw and Gutiérrez 1998, Fullerton and Meekins 2004, Jensen et al. 2004, Seamans et al. 2005, Schmidt 2006, Steger et al. 2006, P. Cranston, L. Diller, S. Farber, G. Gould, J. Keane, B. Marcot, K. Schmidt, M. Seamans, B. Woodbridge, personal communication). Fifty-seven of these 60 detections were in the northern Sierras; the three that were not in the northern Sierras were two hybrid spotted/barred owls in the Eldorado National Forest in the central Sierras (Seamans et al. 2005, M. Seamans, personal communication) and one male barred owl in Kings Canyon National Park in the southern Sierras (Steger et al. 2006). No barred owls have been detected south of San Francisco Bay in western California. If barred owls extend their range farther southward along the coast, the small, isolated groups of CSOs in southern California may be highly susceptible to negative effects from them.

## CONSIDERATION OF CONTROL OF BARRED OWLS FOR CSOS

We agree with the conclusion of the recent 12-month finding (USFWS 2006) that barred owls are

neither a current threat to CSOs nor a threat to CSOs in the foreseeable future. If, just outside of the foreseeable future, barred owls build-up their numbers in the range of the CSO and become a threat, it probably would be too late to manage the problem due to the infeasibility of controlling invasive species after they have become numerous and widespread (Rejmánek and Pitcairn 2002). We view barred owl control for CSOs, if initiated soon, to be an "insurance policy" against barred owls ever becoming a threat to CSOs that also would minimize the numbers of barred owls needing control.

## DRAFT METHODS FOR CONTROL

### Management Options

There are a limited number of ways to keep barred owls from negatively affecting spotted owls, and Buchanan et al. (2007) present five general approaches. Our analysis of these approaches indicates that lethal control is the most practical and effective method (Table 1). Consequently, "control" herein refers to lethal control.

### Control Initiation

Virtually all information we have concerning the effects of barred owls on spotted owls has resulted from incidental detections of barred owls and correlational analyses (Livezey and Fleming 2007). Three ongoing NSO/barred owl radio-telemetry studies in Oregon (D. Rock, D. Weins unpublished data) and Washington (L. Irwin, unpublished data) are attempting to provide direct data. If these studies and at least one control experiment verify that barred owls are indeed the cause of the observed negative effects, then we would have stronger proof on which to justify lethal control of barred owls for CSOs. Consequently, it seems prudent to wait for at least initial results from these studies before initiation of control of barred owls to benefit CSOs.

### Control Locations

A comprehensive program to protect CSOs would require control of barred owls whenever they are found in the range of the CSO. However, politics, public sentiment, and logistics make it unrealistic to assume that barred owls would be

**Table 1.** Comparison of five methods to control effects of barred owls (BDOWs) on spotted owls (SPOWs).

| Method | Known Techniques | Economics | Effectiveness |
|---|---|---|---|
| Habitat management to benefit SPOWs | No | Very expensive; requires large-scale management of SPOW habitat | Unknown |
| Supplementary feeding of SPOWs | Yes | Expensive; requires surveys to locate nesting SPOWs and repeat visits to feed SPOWs | Ineffective; BDOWs could remain and defend their territories from occupation or breeding by SPOWs |
| Diversionary feeding of BDOWs to move them away from areas with SPOWs | Yes | Expensive; requires surveys to locate BDOWs and SPOWs and repeat visits to feed BDOWs | Ineffective; supports reproduction of BDOWs that could later compete with SPOWs |
| Disruption of BDOW reproduction (e.g., oiling of eggs, removal of eggs, sterilization of adults) | Yes | Expensive; oiling or removal of eggs requires surveys to find nests of BDOWs and requires access to nests when eggs are present; sterilization requires capture of BDOWs | Ineffective; BDOWs could remain and defend their territories from occupation or breeding by SPOWs |
| Lethal control of BDOWs | Yes | Moderately expensive; requires locating BDOWs and attracting them to within shooting range | Very effective; eliminates competition with BDOWs |

controlled throughout all forests in the range of the CSO including National Parks, state lands, and private lands. But control may be feasible in National Forest lands, where the vast majority of suitable habitat for spotted owls (and, we assume, barred owls) exists in the Sierras. Hence, control of barred owls only in National Forest lands may be adequate initially. As of 2006, only one barred owl had been detected south of the Eldorado National Forest (Steger et al. 2006), suggesting that control of barred owls in National Forest lands from the Eldorado National Forest northward would be sufficient initially. Agency biologists and volunteer birdwatchers could be enlisted to report detections of barred owls south of Eldorado National Forest and San Francisco Bay to inform decisions concerning expanding control efforts southward. Additional decisions would need to be made concerning control of barred owls in areas outside of National Forests such as in National Parks, state lands, and private lands.

**Detection Techniques**

Barred owls would be detected by eliciting responses using spotted owl calls or barred owl calls produced by voice or by tape recorder/speaker. As in virtually all NSO studies (e.g., Herter and Hicks 2000, Kelly et al. 2003, Olson et al. 2005, Anthony et al. 2006), the studies of CSOs in the Sierras elicit responses from CSOs by using CSO calls. In the two long-term CSO study areas (Plumas-Lassen Administrative Study/Lassen Demography Study, Eldorado Demography Study), we suggest that surveyors continue to use spotted owl calls exclusively, and that barred owls would be detected incidentally. Outside of the CSO study areas, use of barred owl calls rather than CSO calls may be preferred to decrease the chances of aggressive encounters between the species and, possibly, to increase the detection rate of barred owls.

**Control Techniques**

Control of barred owls could be done by individuals (herein called "agents") from a company or agency equipped and permitted to conduct such activity. Barred owls are protected by the Migratory Bird Treaty Act (16 U.S.C. 703 et seq.), so a USFWS permit to lethally control barred owls would be necessary. Barred owls detected in CSO study areas could be reported to agents who would return to the areas in question, elicit responses using barred owl calls, and shoot the responding barred owls. The Principal Investigators of the two CSO long-term study areas in the northern Sierras (J. Keane, personal communication, Plumas-Lassen Administrative Study/Lassen Demography Study, R. Gutiérrez, personal communication, Eldorado Demography Study) are willing to participate in this approach. Outside of these study areas, agents could both survey for and control the barred owls, which would increase efficiency by allowing the agents to control the barred owls when first detected rather than having to relocate them later. USFS conducts many project-specific surveys for CSOs throughout the Sierras, and detections of barred owls from those surveys could be directed to agents. Models of great horned owls (*Bubo virginianus*) (usually used to scare birds from gardens) with their ear tufts cut off could be used to draw barred owls within range. Use of a shotgun from within 30 m of barred owls using 7.5 shot would avoid only injuring barred owls (L. Diller, personal communication).

Other considerations include season and frequency of control, use of carcasses for other research, and coordination with agencies and the public. Season and frequency of survey and control would be dependent on the timing of the nesting season and ease of detection. We suggest that three surveys per 6-month season (March 15-September 15) would be adequate. To accommodate the waning of response as the breeding season progresses, the first survey could be done during the first 6-1/2 weeks (March 15-April 30), the second during the second 6-1/2 weeks (May 1-June 15), and the third during the last 13 weeks (June 16-September 15). Prepared data forms could be used by personnel to record date, location, sex of barred owl, etc. Carcasses could be frozen for use by researchers (e.g., DNA analysis, testing for hybrids). To lessen chances of confrontations with the public, personnel could use unmarked vehicles, and to support their authority if confrontations occur, they could wear uniforms from their company or agency. To lessen cost and, we believe, adequately cover the northern Sierras, a 3-year rotating-panel design could be used to cover the areas to be controlled (J. Keane, personal communication); that is, each of three adjacent areas would be surveyed and receive control annually on a rotating basis. A year-round Project Manager could be in charge of communications with USFS, California Department of Fish and Game, other agencies as needed, game wardens, sheriffs, private landowners, and the public, as well as implementation of the control (e.g., obtaining

permits, training agents, compliance with methods). Good communication and coordination among agencies and the public would be crucial to the success of this endeavor; consequently, we suggest the Project Manager begin these communications at least one year before any control is done. A year-round Contract Manager would be needed to hire and supervise seasonal personnel, administer the contract, and produce annual reports.

**Control Costs**

Cost of control is directly proportional to the area covered. We estimate the average size area that a seasonal agent could cover is approximately 9,200 ha, taking into account varying road densities, area-specific and year-specific differences in snowy and muddy road conditions, snowy and rainy weather during March–May, need for all-terrain vehicles, snow-shoeing, hiking, etc. (C. Gallagher, J. Keane, P. Shaklee, personal communication, D. Rock, personal observation).

**Table 2.** Estimated number of personnel to survey for and control barred owls in suitable habitat in National Forest lands in the northern and central Sierra Nevada Mountains, California.

| Consideration | Area (ha) | No. Personnel |
|---|---|---|
| Area of control | | |
| Total suitable habitat[1,2,3] | 767,027 | |
| Habitat being surveyed via CSO studies[3] | | |
| Lassen Demography Study | 63,380 | |
| Plumas-Lassen Administrative Study | 38,782 | |
| Eldorado Demography Study | 14,535 | |
| Total habitat being surveyed | 116,697 | |
| Remaining habitat requiring surveys | 650,330 | |
| No. of personnel needed to control | | |
| Control only | | |
| Area with CSO surveys | 116,697 | |
| Area one person could cover[4] | 40,000 | |
| No. of areas for control only | | 3 |
| Survey and control | | |
| Area without CSO surveys | 650,330 | |
| Area one person could cover[5] | 9,200 | |
| No. of areas for survey and control | | 71 |
| Total no. of areas for control | | 74 |
| Total no. of personnel needed[6] | | 25 |

[1] The Forest Service considers suitable CSO habitat as forest stands represented by CWHR classes 4M, 4D, 5M, 5D, and 6 (Mayer and Laudenslayer 1988, USDA 2004). USFWS agrees with this classification depending on the structural condition of 4M and 4D stands (USFWS 2006). Here we include all 4M, 4D, 5M, 5D, and 6 forest stands to minimize the chance that we are underestimating areas.

[2] In Lassen, Plumas, Tahoe, and Eldorado National Forests.

[3] U.S. Forest Service data gathered by D. Yasuda.

[4] Assuming 4 times the area that a person could survey and control.

[5] Assuming 15 calling stations/night/surveyor, 3 visits/station/year, 100% coverage of area, 0.5-km hearing circle around each station, 6-month field season (C. Gallagher, P. Shaklee, personal communication, J. Keane, D. Rock, personal observation).

[6] Assuming a 3-year rotating panel design of areas to be controlled (J. Keane, personal communication).

With the methods described here, it appears that a total of 25 seasonal agents and two year-round staff employees would be required to completely cover the area (Table 2). We suggest that the first year be a pilot-study year in which only approximately 10 personnel be used as improvements are made concerning methods of survey and control. Each of these 10 personnel could be placed in one of 3-year panels situated in the northernmost Sierras, where numbers of barred owl are greatest. To minimize effects of barred owls in the CSO study areas and to take advantage of incidental detections of barred owls, three of these 10 personnel could cover the Plumas/Lassen study areas in the northern Sierras and one person could cover the Eldorado study area. Total cost would be dependent on the per-person cost of the company or agency used to do the work. In addition to the costs itemized here, USFWS and USFS would have costs associated with environmental analysis, permitting, administration, and monitoring.

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## LITERATURE CITED

ANTHONY, R. G., E. D. FORSMAN, A. B. FRANKLIN, D. R. ANDERSON, K. P. BURNHAM, G. C. WHITE, C. J. SCHWARZ, J. NICHOLS, J. E. HINES, G. S. OLSON, S. H. ACKERS, S. ANDREWS, B. L. BISWELL, P. C. CARLSON, L. V. DILLER, K. M. DUGGER, K. E. FEHRING, T. L. FLEMING, R. P. GERHARDT, S. A. GREMEL, R. J. GUTIÉRREZ, P. J. HAPPE, D. R. HERTER, J. M. HIGLEY, R. B. HORN, L. L. IRWIN, P. J. LOSCHL, J. A. REID, AND S. G. SOVERN. 2006. Status and trends in demography of northern spotted owls, 1985–2003. Wildlife Monographs 163:1–48.

BECK, T. W., AND G. I. GOULD. 1992. Background and current management situation for the California Spotted Owl. Pages 37–53 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, and T. W. Beck, technical coordinators. The California spotted owl: a technical assessment of its current status. General Technical Report PSW-GTR-133, U.S. Forest Service, Albany, California, USA.

BLAKESLEY, J. A., M. E. SEAMANS, M. M. CONNER, A. B. FRANKLIN, G. C. WHITE, R. J. GUTIÉRREZ, J. E. HINES, J. D. NICHOLS, T. E. MUNTON, D. W. H. SHAW, J. J. KEANE, G. N. STEGER, B. R. NOON, T. L. MCDONALD, AND S. BRITTING. 2006. Demography of the California spotted owl in the Sierra Nevada: report to the U.S. Fish and Wildlife Service on the January 2006 meta-analysis. Colorado State University, Ft. Collins, Colorado, USA.

BUCHANAN, J. B. 2005. Barred Owl. Pages 218–219 *in* T. R. Wahl, B. Tweit, and S. G. Mlodinow, editors. Birds of Washington: status and distribution. Oregon State University Press, Corvallis, Oregon, USA.

BUCHANAN, J. B., R. J. GUTIÉRREZ, R. G. ANTHONY, T. CULLINAN, L. V. DILLER, E. D. FORSMAN, AND A. B. FRANKLIN. 2007. A synopsis of suggested approaches to address potential competitive interactions between Barred Owls (*Strix varia*) and Spotted Owls (*Strix occidentalis*). Biological Invasions 9:679–691.

CAMPBELL, E. C., AND R. W. CAMPBELL. 1984. Status report on spotted owl in Canada, 1983. Report to Committee on the Status of Endangered Wildlife in Canada, Ottawa, Ontario, Canada.

CBD (CENTER FOR BIOLOGICAL DIVERSITY). 2000. Petition to list the California spotted owl (*Strix occidentalis occidentalis)* as threatened or endangered species. April 2000. Portland, Oregon, USA.

CBD (CENTER FOR BIOLOGICAL DIVERSITY). 2004. An updated petition to list the California spotted owl (*Strix occidentalis occidentalis*) as threatened or endangered species. September 2004. Portland, Oregon, USA.

CDFG (CALIFORNIA DEPARTMENT OF FISH AND GAME). 1978. Bird species of special concern in California: an annotated list of declining or vulnerable bird species. California Department of Fish and Game, Sacramento, California, USA.

CHARNLEY, S. 2006. The northwest forest plan as a model for broad-scale ecosystem management: a social perspective. Conservation Biology 20:330–340.

CROZIER, M. L., M. E. SEAMANS, R. J. GUTIÉRREZ, P. J. LOSCHL, R. B. HORN, S. G. SOVERN, AND E. D. FORSMAN. 2006. Does the presence of Barred Owls suppress the calling behavior of Spotted Owls? Condor 108:760–769.

FRANKLIN, A. B., R. J. GUTIÉRREZ, J. D. NICHOLS, M. E. SEAMANS, G. C. WHITE, G. S. ZIMMERMAN, J. E. HINES, T. E. MUNTON, W. S. LAHAYE, J. A. BLAKESLEY, G. N. STEGER, B. R. NOON, D. W. H. SHAW, J. J. KEANE, T. L. MCDONALD, AND S.

BRITTING. 2004. Population dynamics of the California spotted owl (*Strix occidentalis occidentalis*): a meta-analysis. Ornithological Monographs 54.

FULLERTON, S. A., AND D. J. MEEKINS. 2004. Northern Spotted Owl monitoring, banding, and status report on Hawthorne Timber Company ownership. Campbell Timberland Management, Fort Bragg, California, USA.

GREMEL, S. 2005. Factors controlling distribution and demography of northern spotted owls in a reserved landscape. M.S. thesis, University of Washington, Seattle, Washington, USA.

GOVERNMENT OF CANADA. 2002. Species at Risk Act. Government of Canada, Ottawa, Ontario, Canada.

GUTIÉRREZ, R. J. 1994. Changes in the distribution and abundance of spotted owls during the past century. Studies in Avian Biology 15:293–300.

GUTIÉRREZ, R. J., M. CODY, S. COURTNEY, AND D. KENNEDY. 2004. Assessment of the potential threat of the Northern Barred Owl. Pages 1–71 *in* S. Courtney, editor. Scientific evaluation of the status of the Northern Spotted Owl, Sustainable Ecosystems Institute, Portland, Oregon, USA.

HAIG, S. M., T. D. MULLINS, E. D. FORSMAN, P. W. TRAIL, AND L. WENNERBERG. 2004. Genetic identification of Spotted Owls, Barred Owls, and their hybrids: legal implications of hybrid identity. Conservation Biology 18:1347–1357.

HAMER, T. E., E. D. FORSMAN, A. D. FUCHS, AND M. L. WALTERS. 1994. Hybridization between barred and spotted owls. Auk 111:487–492.

HAMER, T. E., D. L. HAYS, C. M. SENGER, AND E. D. FORSMAN. 2001. Diets of Northern Barred Owls and Northern Spotted Owls in an area of sympatry. Journal of Raptor Research 35:221–227.

HAUSLEITNER, D., AND I. R. BLACKBURN. 2006. Inventory and monitoring of the Northern Spotted Owl in British Columbia. British Columbia Ministry of Environment, Victoria, British Columbia, Canada.

HERTER, D. R., AND L. L. HICKS. 2000. Barred Owl and Spotted Owl populations and habitat in the central Cascade Range of Washington. Journal of Raptor Research 34:279–286.

HOBBS, J. 2005. Spotted Owl inventory and nest site descriptions (2004) and telemetry and monitoring of juvenile Spotted Owls (2004 and 2005). British Columbia Ministry of Water, Land and Air Protection, Vancouver, British Columbia, Canada.

JENSEN, H. J., D. B. ADAMS, W. W. MERKLE, AND K. E. FEHRING. 2004. Northern Spotted Owls in Marin County, California. Point Reyes National Seashore, National Park Service, Point Reyes, CA, USA.

JOHNSON, K. N., J. F. FRANKLIN, J. W. THOMAS, AND J. GORDON. 1991. Alternatives for management of late-successional forests of the Pacific Northwest. Report to the Agricultural Committee and the Merchant Marine Committee of the U.S. House of Representatives. Department of Forest Resources, Oregon State University, Corvallis, Oregon, USA.

KELLY, E. G., AND E. D. FORSMAN. 2003. Barred Owl (*Strix varia*). Pages 320–322 *in* D. B. Marshall, M. G. Hunter, and A. L. Contreras, editors. Birds of Oregon: a general reference. Oregon State University, Corvallis, Oregib, USA.

KELLY, E. G, E. D. FORSMAN, AND R. G. ANTHONY. 2003. Are barred owls displacing spotted owls? Condor 105:45–53.

KELLY, E. G., AND E. D. FORSMAN. 2004. Recent records of hybridization between barred owls (*Strix varia*) and northern spotted owls (*Strix occidentalis caurina*). Auk 121:806–810.

LAHAYE, W. S., R. J. GUTIÉRREZ, AND H. R. AKCAKAYA. 1994. Spotted owl metapopulation dynamics in southern California. Journal of Animal Ecology 63:775–785.

LAHAYE, W. S., R. J. GUTIÉRREZ, AND J. R. DUNK. 2001. Natal dispersal of the spotted owl in southern California: dispersal profile of an insular population. Condor 103:691–700.

LESKIW, T., AND R. J. GUTIÉRREZ. 1998. Possible predation of a spotted owl by a barred owl. Western Birds 29:225–226.

LIVEZEY, K. B. 2007. Barred Owl habitat and prey: a review and synthesis of the literature. Journal of Raptor Research 41:177-201.

LIVEZEY, K. B., AND T. L. FLEMING. 2007. Effects of Barred Owls on Spotted Owls: the need for more than incidental detections and correlational analyses. Journal of Raptor Research 41:319-325.

MARCOT, B. G., AND J. W. THOMAS. 1997. Of spotted owls, old growth, and new policies: a history since the Interagency Scientific Committee report. Gen. Tech. Rep. PNW-GTR-408. U.S. Department of Agriculture Forest Service, Pacific Northwest Research Station, Portland, Oregon, USA.

MAYER, K. E., AND W. F. LAUDENSLAYER, JR. 1988. A guide to wildlife habitats of California. California Department of Fish and Game, Sacramento, California, USA.

MAZUR, K. M., AND P. C. JAMES. 2000. Barred Owl (*Strix varia*), No 58 *in* A. Poole and F. Gill, editors. The birds of North America. The Birds of North America, Inc., Philadelphia, Pennsylvania, USA.

NOON, B. R., AND J. A. BLAKESLEY. 2006. Conservation of the Northern Spotted Owl under the Northwest Forest Plan. Conservation Biology 20:288–296.

NOON, B. R., AND K. S. MCKELVEY. 1992. Stability properties of the spotted owl metapopulation in southern California. Pages 187–206 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, and T. W. Beck, technical coordinators. The California Spotted Owl: a technical assessment of its current status. General Technical Report PSW-GTR-133, U.S. Forest Service, Albany, California, USA.

OLSON, G. S., E. M. GLENN, R. G ANTHONY, E. D. FORSMAN, J. A. REID, P. J. LOSCHL, AND W. J. RIPPLE. 2004. Modeling demographic performance of northern spotted owls relative to forest habitat in Oregon. Journal of Wildlife Management 68:1039–1053.

OLSON, G. S., R. G. ANTHONY, E. D. FORSMAN, S. H. ACKERS, P. J. LOSCHL, J. A. REID, K. M. DUGGER, E. M. GLENN, AND W. J. RIPPLE. 2005. Modeling of site occupancy dynamics for northern spotted owls, with emphasis on the effects of barred owls. Journal of Wildlife Management 69:918–932.

PEARSON, R. R., AND K. B. LIVEZEY. 2003. Distribution, numbers, and site characteristics of spotted owls and barred owls in the Cascade Mountains of Washington. Journal of Raptor Research 37:265–276.

PEARSON, R. R., AND K. B. LIVEZEY. 2007. Spotted Owls, Barred Owls, and Late-Successional Reserves. Journal of Raptor Research 41:156–161.

RAND, A. L. 1944. Birds of the Alaska Highway in British Columbia. Canadian Field-Naturalist 58:111–125.

REJMÁNEK, M., AND M. J. PITCAIRN. 2002. When is eradication of exotic pest plants a realistic goal? Pages 249–253 *in* Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Group. IUCN, Gland, Switzerland and Cambridge, United Kingdom.

ROGERS, T. H. 1966. The fall migration: Northern Rocky Mountain-intermountain region. Audubon Field Notes 20:74.

SCHMIDT, K. 2006. Northern Spotted Owl monitoring and inventory, Redwood National and State Parks: 2005 annual report. Redwood National and State Parks, Orick, California, USA.

SEAMANS, M. E., J. CORCORAN, AND A. REX. 2005. Southernmost record of a spotted owl-barred owl hybrid in the Sierra Nevada. Western Birds 35:173–174.

STEGER, G. N., L. R. WERNER, AND T. E. MUNTON. 2006. First documented record of the Barred Owl in the southern Sierra Nevada. Western Birds 37:106–109.

TAYLOR, A. L, AND E. D. FORSMAN. 1976. Recent range expansion of the Barred Owl in western North America, including the first records for Oregon. Condor 78:560–561.

THOMAS, J. W., E. D. FORSMAN, J. B. LINT, E. C. MESLOW, B. R. NOON, AND J. VERNER. 1990. A conservation strategy for the northern spotted owl: a report of the Interagency Scientific Committee to address conservation of the northern spotted owl. U.S. Department of Agriculture Forest Service, Portland, Oregon, USA.

THOMAS, J. W., M. G. RAPHAEL, M. G. ANTHONY, E. D. FORSMAN, A. G. GUNDERSON, R. S. HOLTHAUSEN, B. G. MARCOT, G. H. REEVES, J. R. SEDELL, AND D. M. SOLIS. 1993. Viability assessments and management considerations for species associated with late-succession and old-growth forests of the Pacific Northwest—the report of the Scientific Analysis Team. U.S. Department of Agriculture Forest Service, Washington, D.C., USA.

THOMAS, J. W., J. F. FRANKLIN, J. GORDON, AND K. N. JOHNSON. 2006. The northwest forest plan: origins, components, implementation experience, and suggestions for change. Conservation Biology 20:277–287.

USDA (U.S. DEPARTMENT OF AGRICULTURE). 2004. Sierra Nevada forest plan amendment final environmental impact statement. USDA Forest Service, Pacific Southwest Region, Sacramento, California, USA.

USDA and USDI (U.S. DEPARTMENT OF AGRICULTURE AND U.S. DEPARTMENT OF THE INTERIOR). 1994. Record of decision for amendments to Forest Service and Bureau of Land Management planning documents within the range of the northern spotted owl, and standards and guidelines for management of late-successional and old-growth forest related species within the range of the northern spotted owl. USDA Forest Service and USDI Bureau of Land Management. Portland, Oregon, USA.

USFWS (U.S. FISH AND WILDLIFE SERVICE). 1990. Endangered and threatened wildlife and plants; determination of threatened status for the northern spotted owl; final rule. Federal Register 55:26114–26194.

USFWS (U.S. FISH AND WILDLIFE SERVICE). 1992. Endangered and threatened wildlife and plants; determination of critical habitat for the northern spotted owl; final rule. Federal Register 57:1796–1838.

USFWS (U.S. FISH AND WILDLIFE SERVICE). 1993. Endangered and threatened wildlife and plants; determination of threatened status for the Mexican spotted owl; final rule. Federal Register 58:14248–14271.

USFWS (U.S. FISH AND WILDLIFE SERVICE). 2003. Endangered and threatened wildlife and plants; 12-month finding for a petition to list the California spotted owl (*Strix occidentalis occidentalis*). Federal Register 68:7580–7608.

USFWS (U.S. FISH AND WILDLIFE SERVICE). 2006. Endangered and threatened wildlife and plants; 12-month finding for a petition to list the California spotted owl (*Strix occidentalis occidentalis*) as threatened or endangered. Federal Register 71:29886–29908.

USFWS (U.S. FISH AND WILDLIFE SERVICE). 2007. 2007 draft recovery plan for the Northern Spotted Owl (*Strix occidentalis caurina)*: merged options 1 and 2. U.S. Fish and Wildlife Service, Portland, Oregon, USA.

VERNER, J., K. S. MCKELVEY, B. R. NOON, R. J. GUTIÉRREZ, G. I. GOULD, JR., AND T. W. BECK. 1992. Assessment of the current status of the California spotted owl, with recommendations for management. Pages 3–26 *in* J. Verner, K. S. McKelvey, B. R. Noon, R. J. Gutiérrez, G. I. Gould, and T. W. Beck, technical coordinators. The California Spotted Owl: a technical assessment of its current status. General Technical Report PSW-GTR-133, U.S. Forest Service, Albany, California, USA.

# PRIORITIZATION OF ACTION ON NON-NATIVE SPECIES – PROGRESS IN GREAT BRITAIN

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***Abstract:*** There is an increasing number of calls for action on a wide range of invasive non-native species, but only a limited resource to deal with them. It is clearly important for policy makers and others to be able to prioritize where these scarce resources are directed in order to maximize their conservation benefit. Coordination of effort, and knowledge sharing are clearly important as are risk assessment methods which attempt to quantify the risk posed by a range of species that are either present or are likely to invade. Great Britain, as an island, has advantages over continental landmasses in that prevention of invasion by terrestrial and freshwater species is more straightforward. However, there is still a large number of detrimental non-native species present in the country and this is likely to increase. This paper describes the recent establishment of a mechanism to coordinate action against non-native species in Britain. It also details the development of a risk analysis process which will provide scientifically robust advice to the government and allow policy makers and others to base prioritize actions against invasive species on a more sound footing.

***Key Words:*** coordination, Great Britain, invasive species, prioritization, risk assessment.

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## INTRODUCTION

Recent audits of non-native species for England and Scotland have identified over 2,000 non-native species that occur in the wild (Welch et al. 2000, Hill et al. 2005). A minority of these have seriously detrimental effects and these invasive species probably cost the British economy several billion US dollars per annum (Anonymous 2007). As well as the impact on economic interests such as plant and animal health, there has been a growing recognition in Great Britain (GB) of the significant impact of non-native species to biodiversity interests. This has led to significant developments in relation to dealing with non-native species issues in GB in recent years.

In 2001, the United Kingdom's (UK) Department for Environment, Food and Rural Affairs (Defra) instigated a fundamental review of policy on non-native species issues. This was chaired by Defra and had representatives from the Scottish and Welsh Governments, other relevant government departments and agencies and a range of non-government stakeholders, including conservation and industry representatives. This review identified the single biggest problem in relation to non-native species was the lack of coordinated response across the many stakeholders. The review made a total of 44 recommendations, including eight key recommendations, for improved coordination of response to non-native species in GB:

- Setting up or designating a single co-ordinating body,
- Developing generic risk assessment procedures,
- Improving monitoring and surveillance,
- Developing codes of practice for key sectors,
- Improving capacity for management of non-native species,
- Increasing consultation with stakeholders,
- Improving public awareness, and
- Revising and updating legislation

## COORDINATION

One of the main recommendations of the 2001 review was to designate or establish a single agency or organization to coordinate action on non-native species. In 2005, a Non-native Species Programme Board was established for GB. This Board is chaired by the Defra, and has representatives from the Scottish and Welsh Governments, and other relevant governmental departments and agencies. The Programme Board is the key operational decision-making body on non-native species issues in GB. Its main aims are to oversee the

development of a GB Strategy on invasive species and drive its implementation. It aims to steer and give strategic direction to work undertaken across the government on non-native species and it will establish clear priorities for action. The Programme Board meets approximately quarterly, and to support the work of the Programme Board a separate Non-native Species Secretariat was established in 2006. This Secretariat reports to the Programme Board and has two full-time staff. The Secretariat plays a pivotal role in relation to delivery of actions instigated by the Programme Board, for example establishing working groups to examine specific issues and providing secretarial support for the Risk Analysis Mechanism (see below). It is intended that the Secretariat will become a central hub for information gathering and dissemination concerning invasive non-native species and action being taken to tackle them in GB as well as maintaining links internationally. The Secretariat has recently launched a website www.nonnativespecies.org.

To ensure input from a broad spectrum of expertise, a working group of government and non-government stakeholders has drafted a Framework Strategy for invasive non-native species in GB (see www.nonnativespecies.org/documents/Draft_StrategyV6.4.pdf). This contains 49 specific actions under the headings: Prevention, Detection and Rapid Response, Mitigation and Control, Building Awareness, Legislation, Research and Information Exchange). This strategy has just been through a public consultation process and is being modified in the light of comments received.

## RISK ASSESSMENT

The first research to flow from the recommendations of the 2001 review was in relation to the development of a generic risk assessment methodology to support decision-making and prioritization. It was decided that the generic risk assessment tool would be based on the pest risk analysis (PRA) used by the European and Mediterranean Plant Protection organization (EPPO). The EPPO PRAs follow International Plant Protection Convention standards for pest risk analysis. They are accepted by the World Trade Organization as sufficiently robust from a scientific point of view to justify restricting trade, where appropriate. A consortium of six UK organizations completed a one-year modification of the EPPO PRA scheme with modules to assess the risk of individual species and entry pathways as well as the risk to receptor habitats (www.defra.gov.uk/wildlife-countryside/resprog/findings/non-native-risks/).

The risk assessment works by asking a series of questions that attempt to quantify the probability of entry, establishment and spread, and the magnitude of impact. The response to each question is scored on a five-point scale with a requirement, where possible, to fully justify each response with references. The risk assessment also attempts to quantify any uncertainty by requiring the assessor to indicate the uncertainty of each response on a three-point scale. The results are summarized in three ways:

- The risk rating given by the assessor (based on the opinion of the author),
- A simple summation of the values given for all the risk responses, and
- Using conditional probability to determine a summary score for the risk responses.

The methodology has been peer-reviewed during a separate project (www.nonnativespecies.org/documents/Final_Peer_Review.pdf ), and based on the recommendations from that review, will undergo further development to make it more user-friendly. In the meantime, the existing methodology is being used to assess risk, but only the species and pathway risk assessment modules are being developed and used at present, not other modules. Each risk assessment is carried out by one expert in the taxon/pathway being assessed. To ensure that it is an accurate reflection of the current state of knowledge on the species/pathway concerned, a Risk Analysis Mechanism which essentially reviews the assessment in a similar way to the peer review and editorial processes for scientific papers was developed. All risk assessments are first reviewed by a single peer reviewer, who is also an expert in the taxon/pathway concerned, who ensures that the literature quoted is appropriate, correctly interpreted, and no important literature is omitted. The risk assessment plus the peer reviewer's comments are then reviewed by a panel of risk assessment experts to ensure accuracy of approach and correct application of the methodology. Comments from the panel and peer reviewer are sent to the original risk assessor to address. When the assessment is finally signed off by the risk analysis panel it is sent to the Programme Board to help inform their actions. A fast-track mechanism has also been designed to ensure that the risk

assessment process helps with rapid response rather than impede it.

## MONITORING AND SURVEILLANCE

It is extremely important to have accurate data on detections of new non-native species as well as to monitor changes in existing non-native species distributions. GB generally has excellent biological recording mechanisms, but for some taxa recording non-native species is seen as being of low priority. We are currently developing procedures, to improve recording of non-native species and to increase the speed of reporting interceptions of highly detrimental species. This information will be used to trigger risk assessments and potentially to initiate a rapid response to eradicate problem species before they become established. Rapid response actions are currently being carried out for American bullfrogs (*Rana catesbeiana*) and water primrose (*Ludwigia grandiflora/peploides*) in England.

## CONCLUSION

Great Britain has been making substantial strides in recent years in relation to action against non-native species. Britain has achieved some notable eradication successes in the past, but there is now a shift towards a more proactive and preventative approach to tackling invasive species. We currently have a substantial proportion of the necessary components in place to enable an effective response against invasive species. Priorities for the near future include establishing a comprehensive rapid response mechanism, establishing an education and awareness working group to develop a communications and media relations strategy, and strategy coordinating the control of existing non-native species.

## LITERATURE CITED


ANONYMOUS. 2007. The invasive non-native species framework strategy for Great Britain. Department for Environment, Food and Rural Affairs, London.

HILL, M., R. BAKER, G. BROAD, P. J. CHANDLER, G. H. COPP, J. ELLIS, D. JONES, C. HOYLAND, I. LAING, M. LONGSHAW, N. MOORE, D. PARROTT, D. PEARMAN, C. PRESTON, R. M. SMITH, AND R. WATERS. 2005. Audit of non-native species in England. English nature research report. Peterboro 662:1-81.

WELCH, D., D. N. CARSS, J. GORNALL, S. J. MANCHESTER, M. MARQUISS, C. D. PRESTON, M. G. TELFER, AND H. ARNOLD. 2000. An audit of alien species in Scotland. The Institute of Terrestrial Ecology, Banchory, Scottland.

# THE FAILED REGULATORY SYSTEM FOR ANIMAL IMPORTS INTO THE UNITED STATES – AND HOW TO FIX IT

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***Abstract:*** This paper provides a synopsis of the Defenders of Wildlife report entitled *Broken Screens: The Regulation of Live Animal Imports in the United States*, released in August 2007. That report assessed the complex federal system for regulating live wild animal imports as applied to the 2,241 non-native species that were identified in United States Fish and Wildlife Service records as being imported between 2000 and 2004, inclusive. The report describes the "coarse risk screening" conducted for those species by searching the scientific literature and United States and international databases. If one or more reliable sources indicated a species was known or predicted to be invasive, pose a disease risk, or otherwise be harmful, in the United States or elsewhere, the species was labeled "potentially risky." Due to the hundreds of potentially risky species being imported with no-risk screening by federal officials, the report concludes the United States' regulatory system provides a low level of protection to the nation. Absent major policy reforms, some of the potentially risky, imported animals will escape or be released and form invasive wild populations or cause disease outbreaks. Eleven policy recommendations are offered to reduce these risks to more tolerable levels.

***Key Words:*** animals, disease, federal regulations, imports, invasive species, laws, non-native, risk, screening.

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## INTRODUCTION

The United States (US) is the world's largest importer of live wild animals, receiving hundreds of millions each year, most destined for the pet and aquarium trade, some for zoos, research labs, and specialty food markets. The declared wholesale value of these imports from 2000 to 2004 was more than one-half billion dollars, but this lucrative import industry is a risky business. Inevitably, some of the animals escape or, no longer wanted, are let go. Unchecked by natural controls such as predators and pathogens found in their native ranges, these species can spread and cause serious environmental, health, and economic problems. Left unchecked by the federal agencies with regulatory authority over this trade, potentially risky species continually enter the US.

Remarkably, none of these federal agencies do comprehensive risk screening of these species before they are allowed into the US. Moreover, the species in trade are often not adequately documented in public records. Like a set of "broken screens," the complex federal system for regulating live, wild animal imports is ineffective at keeping out harmful or potentially harmful, species. Absent major policy reforms, some of the potentially risky non-native animals imported into this country will escape or be released and establish invasive wild populations or cause disease outbreaks.

Recognizing the failings of the regulatory system and the lack of data necessary to understand and fix the broken screens at the national borders, Defenders of Wildlife (herein after, Defenders) prepared an unprecedented study to: (1) describe the full scope of the trade -- what non-native species are being imported and in what cumulative quantities, (2) assess the risks of the trade -- the impacts these animals are having or could have on native species, U.S. lands and waters, and on human health, (3) examine the broken screens -- the federal system for regulating this trade, and detail their weaknesses and inconsistencies, and (4) make policy recommendations based on these findings.

The Defenders' study, *Broken Screens: The Regulation of Live Animal Imports in the United States*, was released in August, 2007 (Defenders of Wildlife 2007). The report and extensive supporting information are available online at: www.defenders.org/animalimports. This paper is a synopsis of that fully-referenced report.

## SCOPE AND RISKS OF THE LIVE WILD ANIMAL TRADE

Defenders reviewed summaries of the 2000-2004 records kept by the US Fish and Wildlife Service (FWS) agents at ports of entry and gathered information never before collated, including species names for the 2,241 species of non-native, wild animals legally imported to the US. Working with the Consortium for Conservation Medicine (CCM) and the World Conservation Union (IUCN) Invasive Species Specialist Group (ISSG), Defenders coarsely screened all these species for potential risks by searching the scientific literature and online databases and canvassing recognized invasive species experts. If one or more of these reliable sources indicated a species was known or predicted to be invasive or harmful in the US or elsewhere, the species was labeled potentially risky.

According to the readily accessible scientific evidence gathered by Defenders and the ISSG, 302 of the 2,241 animal species imported to the US, 13 percent, were classified as potentially risky (Table 1). Sixteen percent of vertebrates were classified as potentially risky – mostly birds, fish and mammals. Although these species primarily presented as invasive risk, human and non-livestock disease risks also were prevalent. Only 3 percent of the identified invertebrates were tagged potentially risky, but this group of animals is relatively unstudied compared to vertebrates.

## IMPACTS ON ENVIRONMENT AND HEALTH

Invasive, non-native species can eat, outcompete, parasitize, and transmit diseases to native wildlife. They also can alter the physical environment, modifying or destroying habitat important to native wildlife. Particularly on islands and in isolated lakes and springs, invaders have done extensive environmental damage. In Hawaii, for example, the egg-eating Indian mongoose (*Herpestes javanicus*) has driven rare native birds toward extinction. Indeed, invasive species are a commonly cited contributing factor in listings under the federal Endangered Species Act. Hundreds of invasive species, including at least 26 considered by experts to be of "high impact," are already established in the US. Several more, including Burmese pythons (*Python molarus bivittatus*) in the Florida Everglades, red lionfish (*Pterois volitans*) in the western Atlantic and suckermouth catfish (*Hypostomus plecostomus*) in the waters of the South, are gaining a foothold. Many other likely invaders lurk among the animals currently in trade. Imported animals can also carry infectious pathogens and harmful parasites.

Although the US is the top market for live animal imports, the federal government does not require most imported animals (with the exception of commercially-imported birds, livestock, and a

**Table 1.** Proportion of identified non-native animal species imported into the US with annotations for potential invasiveness or disease risk, 2000-2004.

| Taxonomic Group | Total Imported Non-native Species | Non-native Species with Risk Annotations | Proportion of Non-native Species with Risk Annotations |
|---|---|---|---|
| Vertebrates | | | |
| Amphibians | 172 | 13 | 8 % |
| Birds | 559 | 129 | 23 % |
| Fish | 121 | 36 | 30 % |
| Mammals | 263 | 61 | 23 % |
| Reptiles | 710 | 52 | 7 % |
| Total vertebrates | 1,825 | 291 | 16 % |
| Total all invertebrates | 416 | 11 | 3 % |
| Grand total | 2,241 species | 302 species | 13 % |

**Table 2.** Infectious agents introduced to the US via imports of live, wild animals, 1996-2006.

| Infectious Agent | Most Recent Documentation | Imported Host | Known Carrier Hosts | Infected Animals | Status in US |
|---|---|---|---|---|---|
| Exotic Newcastle's Disease | 1999 | Various avian species | Various avian species | Poultry | Localized, recurring outbreaks |
| Heartwater | 2000 | African tortoise tick | lizards, snakes, and tortoises | Domestic livestock, white-tailed deer | Present |
| Malignant Catarrhal Fever | 2002 | Ankoli cattle | Wildebeest | Ruminant species | Eradicated |
| Monkeypox Virus | 2003 | Gambian giant pouched rats | Gambian giant pouched rats | Humans, prairie dogs | Eradicated |
| Viral Hemorrhagic Disease of Rabbits | 2005 | European rabbit | European Rabbit | European rabbit | Recurring localized outbreaks |
| Chytridiomycosis | 2006 | American bullfrog | American bullfrog, African clawed frog | Amphibians | Present |
| Ranavirus | 2006 | American bullfrog | American bullfrog | Amphibians | Present |

few other animals) to undergo a quarantine period or to have proof of veterinary clearance from their country of origin. This relaxed approach leaves the US open to "pathogen pollution," the human-driven introduction of various infectious agents to new locations (Table 2). In the current globalized marketplace, shipping and selling live animals with minimal regulation magnifies the risks to public health, animal health, and the economy. Examples include the 2003-2004 outbreak of SARS (severe acute respiratory syndrome) traced to the sprawling, unsanitary live animal markets of southern China, and monkeypox transmitted by prairie dogs (*Cynomys ludovicianus*) sold as pets that contracted it from Gambian giant pouched rats (*Cricetomys gambianus*) imported from Africa by a US pet distributor.

Now, a particularly virulent strain of avian influenza (AI), which has turned up in wild-caught birds imported into Europe, has the potential to spread to the US. This strain of AI poses a global pandemic threat should the virus evolve the ability to spread more efficiently among humans. Millions of people could die and worldwide economic damage could be $200 billion or more. The US import trade in live animals that are potential AI carriers, if not better regulated, could be a major factor in a future US outbreak.

Global warming is likely to intensify the threats from harmful species invasions and diseases. Tropical species will acclimate better and be able to survive winters that once killed them with cold temperatures. New invaders may include hot climate vectors of West Nile virus and other destructive human and animal pathogens.

## BROKEN SCREENS: THE SYSTEM FOR REGULATING WILD ANIMAL IMPORTS

The US legal system for live animal imports generally gives authority to several agencies:

- FWS, under the Lacey Act to prohibit specified "injurious" animals;
- The U.S. Department of Agriculture's Animal and Plant Health Inspection Service (APHIS), under the Animal Health Protection Act and Plant Protection Act, allowing, but not requiring, APHIS to regulate any animal that is a pest or disease carrier threatening to farmed livestock or any plant;
- The Department of Health and Human Service's Centers for Disease Control and Prevention (CDC), under the Public Health Service Act, allowing, but not requiring, CDC to regulate any animal posing a human disease risk.

The Lacey Act, a law enacted in 1900 that is in desperate need of an overhaul, is based on a listing approach that is excruciatingly slow, offers minimal coverage and is repeatedly criticized as outdated and ineffective by experts. The interagency National Invasive Species Council proposed a plan in 2001 to fill some of the import screening gaps, but has failed to follow through on it. APHIS's approach to protecting plants and domesticated livestock from harmful pests is stronger and more expedient than FWS's Lacey Act authority, but it focuses on just a narrow segment of risks posed by the animal import trade. CDC's statutory authority is broad, but is wielded reactively and minimally. For example, after the 2003 monkeypox outbreak caused by imported Gambian rats, CDC claimed it would be more aggressive in protecting public health, but it has not been.

In summary, FWS and CDC regulatory laws currently provide the nation a low level of protection from potentially invasive or "injurious" species and from species that pose infectious disease risks to humans, non-livestock animals or both.

## MENDING THE SCREENS: POLICY RECOMMENDATIONS

Defenders urges immediate implementation of 11 policy changes to mend the broken screens at the US national borders and vastly reduce the risks of the live animal import trade.

1. Pass new national legislation that clearly directs FWS, CDC and all other federal agencies to follow a more risk-averse national standard for wild animal imports. Defenders recommends the following language should be the standard: "Federal agencies shall only allow imports and interstate commerce in non-native animals that have been assessed by a responsible federal official and determined to pose a low likelihood of causing harm to the environment, the economy, public health or animal or plant health in the United States."

2. Amend the Lacey Act to direct FWS to conduct detailed pre-import screening of live, wild animals and give FWS authority to provisionally prohibit any species for which adequate scientific information is not available.

3. Promptly and fully analyze the risks of continuing to import the 302 species identified by Defenders and the ISSG as potentially risky. (Defenders of Wildlife 2007).

4. Aggressively enforce the existing federal regulation (50 CFR 14.53) that full species identification must accompany every animal shipment, and make that information available to the public.

5. Immediately address the threat of AI by rigidly enforcing the 30-day quarantine required for all bird imports and testing every bird.

6. Coordinate and strengthen the federal government's role in overseeing and regulating health risks associated with live animal imports.

7. Appropriate more funds for research on diseases transmitted by imported animals.

8. Implement the post-import recommendations adopted by the National Association of State Public Health Veterinarians and the Council of State and Territorial Epidemiologists (at www.cste.org/ps/2003pdfs/03-id-13%20%20final.pdf ).

9. Increase funding and staffing for FWS port inspectors, the country's first line of protection from illegal and diseased animal imports.

10. Implement an application fee system to pay for the bulk of the cost of pre-import screening of live wild animals.

11. Include the animal import industry and other stakeholders in the development of policy solutions.

Harmful new invasions and threatening diseases must be stopped from entering the US. The federal government could readily enforce stricter controls because there are only a few dozen major shipping ports, airports and border crossings in the US where legal animal imports enter. The 11 recommendations presented here, supported by the new information on the animal import trade assembled by Defenders, CCM and the ISSG, show the way to fixing the nation's broken screens.

## LITERATURE CITED

DEFENDERS OF WILDLIFE. 2007. Broken screens: the regulation of live animal imports in the United States. Report by Defenders of Wildlife, Washington, D.C., USA. (Fully-referenced report and extensive background data and related information are available online at: www.defenders.org/animalimports.).

# MONITORING THE SALE AND TRAFFICKING OF INVASIVE VERTEBRATE SPECIES USING AUTOMATED INTERNET SEARCH AND SURVEILLANCE TOOLS

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***Abstract:*** The internet is often an unregulated pathway that can lead to the introduction of invasive species. Surveillance of internet-related sale or trade activities can reduce the movement of potentially invasive species and serve to lessen the risk of an introduction. The Invasive Species Internet Monitoring System automates the process of searching the internet for suspect sites (e.g., internet storefronts, chat rooms, etc.) involved in the sale or trade of targeted species. Additionally, this software system incorporates data management features that permit the archiving of suspect website content, assists subject matter experts in evaluating sites for risk, send informational emails to website owners and can track sites for compliance. This software system is currently being used by United States Department of Agriculture, Animal and Plant Health Inspection Service, Plant Protection and Quarantine and the Smuggling, Interdiction and Trade Compliance Program to search for websites selling regulated insects, weeds, mollusks, fruits and vegetables, and for animals and animal products that have the potential to carry Highly Pathogenic Avian Influenza.

***Key Words:*** automated search, internet, invasive species, search engine, surveillance.

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## INTRODUCTION

The introduction of alien invasive species, either accidentally or purposefully, into non-native habitats poses serious threats to native ecosystems and to human and animal health. The establishment of invasive species has been directly linked to reduced native biodiversity and habitat loss (e.g., D'Antonio and Vitousek 1992). Additionally, the introduction of monkey pox virus in native prairie dog populations from imported Gambian giant pouch rats ultimately led to the infection of humans in the United States (US, Pacelle 2003). Identifying the risks and pathways that can lead to the introduction of invasive species and transmittable zoonotic pathogens can serve to reduce the likelihood of their establishment in new habitats and populations (Kasari 2006).

As early as 1999, it was clear that the US Department of Agriculture (USDA), Animal and Plant Health Inspection Service's (APHIS) regulated organisms were being sold on the internet. Ebay.com, internet discussion and interest groups, chat rooms, etc., all play a part in these sale and trade activities. This mode of illegal plant and animal trade was identified as a new pathway, that unless closed, could lead to the introduction of new alien pests and pathogens into the US. In 2002, the Invasive Species Internet Monitoring System (ISIMS) was developed as a cooperative effort between the USDA and the National Science Foundation (NSF) Center for Integrated Pest Management (CIPM). The goal of the project was to develop techniques and tools that would semi-automate the process of searching for invasive pests being sold on the internet. The integration of automated search tools (e.g., FAST Search and Transfer, Inc.) in combination with custom analysis and databasing tools (e.g., ISIMS), can aid in monitoring the sale and trafficking of invasive vertebrate species. FAST Search is a web crawler that can locate websites selling or trafficking federally regulated invasive species or other organism or animal products of interest. ISIMS is a web application that aids analysts in evaluating, investigating and monitoring websites that may offer invasive vertebrate species for sale or trade.

## METHODS

### Search Engine

The core component is the FAST Search and Transfer Web Search Engine. This application automates the task of searching the web (all published web pages, news feeds, etc.) using key terms that define distinct query parameters in order to return relevant search results. CIPM has highly customized this application and search pipeline in

order to facilitate searching for invasive species information.

**Input Parameters**

The list of key search terms typically consists of a list of invasive species (scientific names and common name synonyms). Connector terms that define a sale action on a website are identified (e.g., "for sale", "Mastercard", "shopping cart"), while a list of exclusion terms (e.g., "quarantine", "toy", "www.kfc.com") that remove sites from consideration is also generated. Lists of websites that should always be searched (e.g., "www.invasivespeciesforsale.com") or those that should be excluded from the automated search (e.g., "discovery.com") are created, and the user has the ability to weight or load specific search criteria in order to boost search relevancy. The data is then stored in tables in a Structured Query Language Server database.

**Search Details**

Scripts automate the creation of the search query strings based on the input parameters provided. Once the query strings are generated and stored in the database, the FAST search engine initiates the search of the indexed web universe for documents matching the search criteria. Once a search is initiated, the FAST search engine continually queries the web for new documents that match the search criteria.

The CIPM utilization of FAST Search is configured to search through web documents, news feeds, blogs, in addition to chat rooms, if accessible. Over sixty languages can be queried in order to search for colloquial words and phases on foreign websites.

**Identifying Websites of Interest**

A customized web interface is used to display the results of each search. Subject Matter Experts (SME's) review the results (website pages) to determine if a site is selling an invasive species of interest. Sites identified by SME's as relevant are saved to a database for further analysis. The system incorporates a Site Evaluation Tool that permits an evaluator to examine a website in detail. Integrated tools such as a "WHOIS" search assist evaluators in identifying the location and owner of websites that may be selling invasive species. Enforcement officers have the ability to send informative letters via email to potential violators. All information pertaining to a potential violator, in addition to any letters and associated documents related to a site, are stored for reference.

**Site Analysis**

Summary statistics and reposts can be generated in order to estimate the efficacy of and evaluate the parameters used in each search. These tools permit refining of search parameters in order to generate more relevant search results.

## RESULTS

Query results are returned to a custom data view (Figure 1). The results shown are a subset of the species listed on the U.S. Fish and Wildlife Service Threatened and Endangered Animals and Plants list.

Results are ranked in order of their relevancy to the number of search terms matched in the query. Selecting the page title link will bring up the original web page. The contact information of the web page owner or host site can be viewed by selecting the "getWhoIS" link. Sites can be marked as relevant and saved for further analysis or irrelevant. If sites are returned that are not of interest, they can be excluded from the search result list. Additional site filters can be used to return only those sites that correspond to a particular location, language, or indexed keyword.

Utilizing this methodology and the website, we located the GiantPouchedRatz Yahoo discussion group. This demonstrated that Gambian giant pouched rats are still being bred and offered for sale in the US even though now illegal (Figure 2).

## DISCUSSION

By automating the search process, the ISIMS software can identify websites selling suspect invasive vertebrate species and other species of interest. However, in order to return relevant search results, the query parameters for each specific collection of organisms must be well defined. The input of subject matter experts when developing query parameters is essential to the success of any search. Additionally, the commitment of resources towards reviewing query results and determining if sales or trade activity warrants further action is also a crucial component of this system.

The focus of much of the work we have performed to date has been to identify websites offering regulated organisms for sale or trade via the internet. However, the types of information

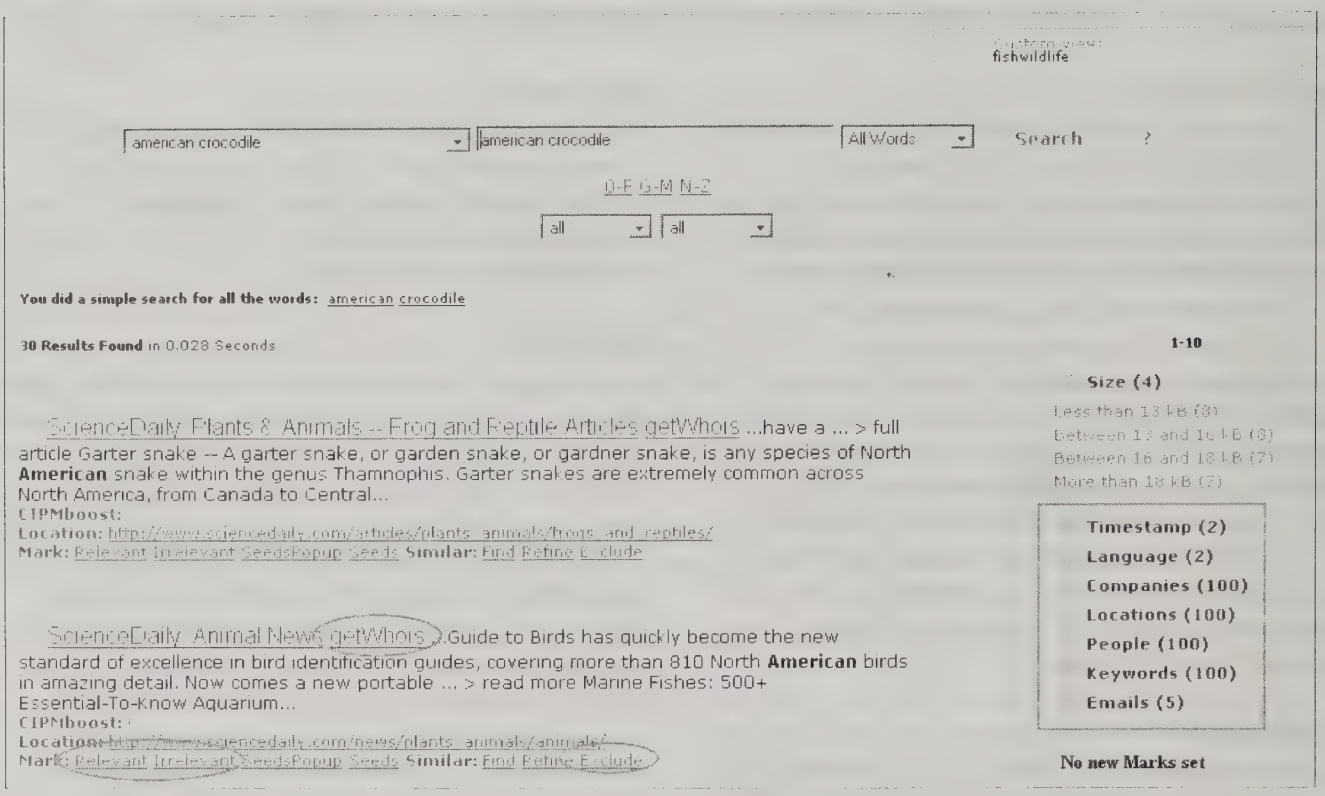



**Figure 1.** An example of data output from the FAST search engine in a custom data view.

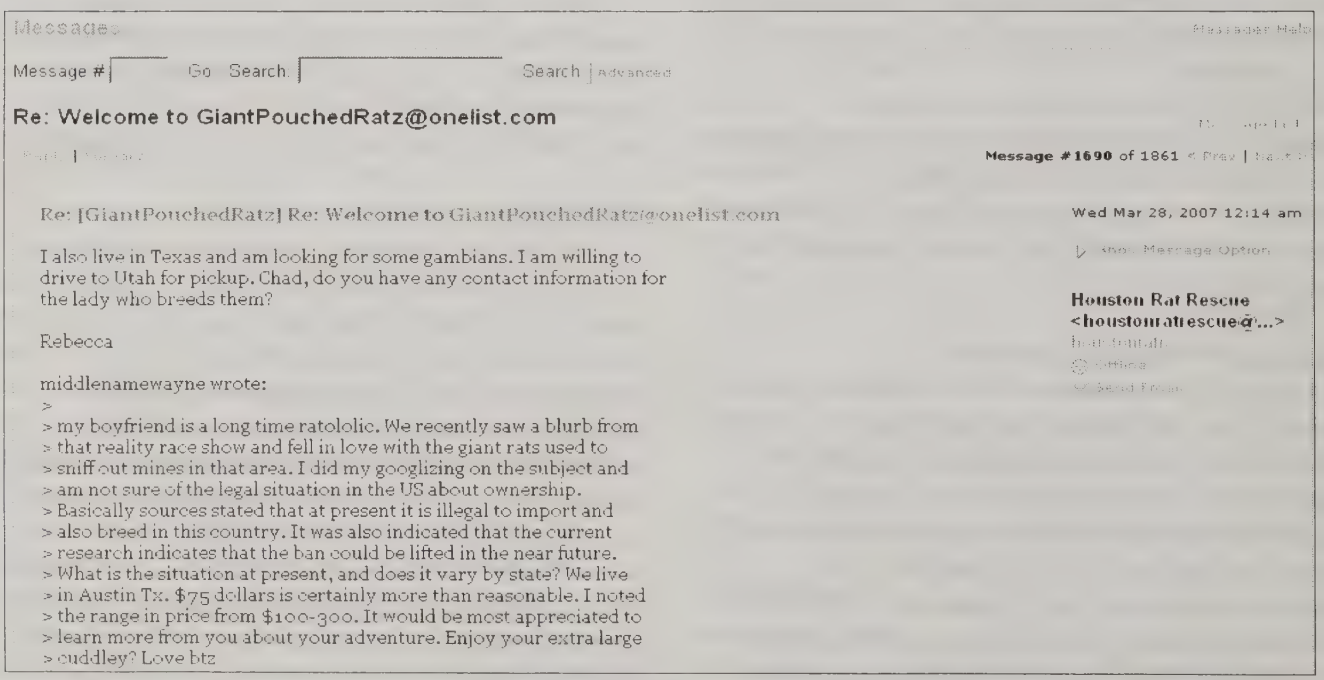



**Figure 2.** Chat group message indicating that Gambian giant pouched rats are currently being bred and sold in the US.

returned from ISIMS queries can be adapted for many different needs and product groups:

- Identifying sites selling regulated organisms or species of interest,
- Sites selling only within a state or political region,
- Foreign websites selling invasive species,
- Provide a list of growers/producers within a state or region, and
- Internet sales of Convention on International Trade in Endangered Species listed species

One example of this alternate use was identified during a search for websites selling avian products that could serve as a vector for Highly Pathogenic Avian Influenza. After numerous searches and refinements of search terms, it was concluded that few producers and suppliers of avian products, especially those from southeast Asian markets and the US, advertised their products via internet commerce sites. Thus, focus was shifted from identifying sites selling avian products to identifying those overseas suppliers that advertise that they are willing to ship avian products worldwide. The APHIS' Smuggling, Interdiction and Trade Compliance Program is working with other groups to determine if data from these sites can be used to assist in targeting suspect shipments from overseas suppliers.

## LITERAURE CITED


D'Antonio, C. M. and Vitousek, P. M. 1992. Biological invasions by exotic grasses, the grass/fire cycle, and global change. Annual Review of Ecology and Systematics 23:63–87.

Kasari, T. 2006. Risk analysis team helps guide surveillance plans through pathways analysis. Risk Assessment. NAHSS Outlook Quarter Two. (www.aphis.usda.gov/vs/ceah/ncahs/nsu/outlook).

Pacelle, W. 2003. Importation of exotic species. U.S. Senate Committee on Environment & Public Works Hearing Statements. (http://epw.senate.gov/hearing_statements.cfm?id=212888).

# USING PATHWAY ANALYSIS TO INFORM PREVENTION STRATEGIES FOR ALIEN REPTILES AND AMPHIBIANS

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***Abstract:*** Alien reptiles and amphibians are deserving of greater attention that has hitherto been bestowed upon them by managers and researchers. Eradication or control of established taxa will generally be infeasible, leaving prevention of introductions as the primary management tool for controlling herpetological invasions. I analyzed >5,700 introductions of alien reptiles and amphibians worldwide to obtain the pathway information necessary for design of informed prevention programs. Six pathways account for the large majority of introductions: accidental introductions via cargo and the nursery plant trade and intentional introductions for biocontrol, food use, the pet trade, and aesthetic purposes. Pathway importance varies taxonomically, temporally, and geographically. Unlike other taxa for which introductions have been dominated by either accidental pathways alone or intentional pathways alone, reptile and amphibian introductions involve a mix of both. Consequently, prevention programs must involve a two-pronged approach for these taxa: risk assessment of pathways for taxa introduced accidentally and risk assessment of species for taxa introduced intentionally. Because of variation in pathway importance, information on how taxonomic, temporal, and geographic variables co-vary with economic and social data may allow for predictive assessment of pathway risk for accidental introductions. In contrast, some predictive assessment of taxon risk was achieved using variables that measure climate-matching between native and introduced ranges, phylogenetic risk, and prior history of successful taxon establishment.

***Key Words:*** alien species, amphibians, importation, invasive species, nursery trade, pet trade, reptiles, risk assessment.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

In the past decade, invasive alien species have begun to receive (outside the Austral English-speaking countries, which have long been responsive to the problem) the increased scientific and managerial attention they merit by virtue of their ecological and economic destructiveness. For terrestrial invasive species, this attention has a marked taxonomic bias, with considerable study and management directed toward mammals, plants, and economically important insects, and less so toward other taxa. This is reasonable inasmuch as those priority taxa cause tremendous damage that is frequently apparent even to the casual observer. The concern, though, is that silence about the ignored taxa not be mistaken for a tacit admission of ecological inconsequentialness.

Among those alien taxa largely overlooked as scientific or management subjects are reptiles and amphibians, collectively referred to as "herps". With the exceptions of brown treesnakes (*Boiga irregularis*) in Guam, bullfrogs (*Rana catesbeiana*) in the western United States (US), and cane toads (*Bufo marinus*) in Australia, sustained efforts to assess the ecological impacts of alien populations of amphibians or reptiles are lacking. However, impacts have been reported for a number of other alien herp populations (Kraus, in prep.) and descriptive autecological data are available for a modest number of additional populations. This is a remarkable degree of inattention given that 1,034 naturalizations of 315 species of alien herps have successfully established in 287 political jurisdictions worldwide. The reasons for this negligence are likely twofold. First, most alien reptiles and amphibians do not directly affect humans, their domestic stock, or crops. Indeed, their effects are likely to be nuanced ecological impacts on other wildlife (e.g., insects, other herps) whose ecological importance is readily discounted by most humans. Hence, they are easily perceived – including by most scientists – as of little concern. Second, most species, even large ones, are cryptic. This makes them difficult for the average human to notice and difficult for any scientist to study. Their study is made more difficult by the compounding

fact of their nuanced ecological effects, which often takes careful experimental research to determine. Consequently, alien herps rarely give the impression of occurring at plague proportions, as do more visible taxa.

In this paper, I briefly review evidence suggesting that this frequent oversight of alien herps is neither warranted nor advisable. This evidence is presented in greater detail in a forthcoming book (Kraus, in prep.), and I will give only the most cursory summary of that evidence here. I then examine what mitigation measures might be feasible for alien reptiles and amphibians and how information derived from analysis of the pathways by which these animals are transported can potentially assist in the design of effective measures to prevent future invasions. That work is ongoing, so this paper largely represents a broad conceptual overview of how alien herp invasions might best be mitigated and a short status report of efforts to date to meet that goal.

## MATERIALS AND METHODS

I reviewed the literature up through the end of 2005 to construct a database of alien reptile and amphibian introductions worldwide. Data collected, when available, included jurisdiction to which introduced, date(s) of introduction, pathway(s) of introduction, minimum number of introductions involved, and whether the introduction led to a currently established population. Jurisdictions were typically countries but distinct island groups were tracked separately from the remainder of the country (e.g., Galapagos separate from Ecuador, Ryukyu and Ogasawara archipelagos separate from Japan) and species were tracked by state and province within the US and Canada. From these data, analyses were conducted to assess the pathways by which herp species were being transported and to determine how those pathways have varied. The same literature was reviewed to assess what ecological and economic impacts have been reported for alien herp introductions and what control measures have been attempted to date.

## DO ALIEN REPTILES AND AMPHIBIANS WARRANT BEING IGNORED?

A variety of ecological and social impacts from alien reptiles and amphibians has been documented, even though only a small pool of species has been studied rigorously. Documented ecological impacts include predation or poisoning of native species, secondary trophic effects, competition, and vectoring of parasites; evolutionary effects include genetic contamination via hybridization, as well as changes to morphology, physiology, and behavior (Table 1). A number of these impacts directly affect endangered native species, with perhaps the most dramatic example affecting Raffone's wall lizard (*Podarcis raffonei*), which is almost extinct due to competition (and perhaps genetic swamping) from introduced Italian wall lizard (*P. sicula*; Capula et al. 2002). Direct effects on humans include economic losses to agriculture, power supplies, and property values; health impacts via envenomation, water contamination, and disease vectoring; and loss of scientific knowledge in biogeography, taxonomy, and ecology (Table 1). In short, alien reptiles and amphibians are capable of causing many of the negative impacts widely appreciated in better-known taxa. Yet, despite this broad array of documented effects, ecological impacts have been little studied in alien herps, being noted for only 23 of 315 species having established extra-limital populations. And non-epistemological effects on humans are reported for only six of these populations (Table 1). Loss of scientific knowledge has been a fairly common result and only some of the more obvious examples are provided in Table 1, although additional examples from Mediterranean, Caribbean, and Southeast Asian islands likely exist. It is to be expected that damaging impacts are far more widespread than currently evident and that examples will multiply once a broader array of taxa and populations is sampled.

Establishment of jurisdictional populations of alien reptiles and amphibians has been growing exponentially since 1850 (Figure 1), with a doubling time of 30.4 years (Kraus, in prep.). Thus, introductions are not abating, and the pool of invasive herp species is likely to grow in the foreseeable future. The wide array of impacts already documented despite sparse study, coupled with the rapidly increasing pace of alien herp naturalization, lead me to conclude that negligence of alien reptiles and amphibians is not warranted or advisable from either an ecological or purely anthropocentric viewpoint. Instead, I suggest that greater scientific and managerial attention to these taxa, their impacts, and potential tools for their management is overdue.

**Table 1.** Impacts reported for alien reptiles and amphibians. Only ecological impacts on natives are noted; species reported to affect other aliens are omitted. Only a few of many possible examples are given of the scientific impacts. Data taken from Kraus (in prep.), which details references for all instances.

| Impact | Species |
|---|---|
| Ecological | |
| Predation | *Anolis carolinensis, Anolis sagrei, Boiga irregularis, Bufo marinus, Natrix maura, Osteopilus septentrionalis, Rana catesbeiana, Xenopus laevis* |
| Poisoning | *Bufo marinus* |
| Secondary trophic effects | *Anolis carolinensis, Boiga irregularis, Bufo marinus, Carlia ailanpilai, Hemidactylus frenatus, Rana perezi* |
| Competition | *Anolis carolinensis, Anolis sagrei, Bufo marinus, Carlia ailanpilai, Eleutherodactylus johnstonei, Hemidactylus frenatus, Osteopilus septentrionalis, Podarcis sicula, Rana catesbeiana, Trachemys scripta* |
| Disease vector | *Ambystom tigrinum, Rana catesbeiana, Xenopus laevis* |
| Evolutionary | |
| Hybridization | *Ambystom tigrinum, Anolis distichus, Anolis sagrei, Cuora flavomarginata, Emys orbicularis, Iguana iguana, Podarcis sicula, Protobothrops elegans, Rana esculenta, Rana lessonae, Rana ridibunda, Sauromalus spp., Trachemys scripta, Triturus carnifex* |
| Changed morphology | *Bufo marinus* |
| Changed physiology | *Bufo marinus* |
| Changed behavior | *Bufo marinus, Natrix maura, Rana catesbeiana* |
| Economic | |
| Agriculture | *Boiga irregularis, Bufo marinus, Varanus indicus* |
| Power supplies | *Boiga irregularis* |
| Property values | *Eleutherodactylus coqui* |
| Human Health | |
| Envenomation | *Boiga irregularis* |
| Water contamination | *Bufo marinus* |
| Disease vector | *Bufo marinus, Eleutherodactylus johnstonei* |
| Airstrike hazard | *Iguana iguana* |
| Scientific | |
| Biogeography | *Cryptoblepharus poecilopleurus, Eleutherodactylus johnstonei, Eleutherodactylus martinicensis, Emoia cyanura, Emoia impar, Gehyra mutilata, Gehyra oceanica, Geochelone carbonaria, Hemidactylus garnotii, Hemiphyllodactylus typus, Iguana iguana, Lepidodactylus lugubris, Lipinia noctua, Nactus pelagicus, Rana ridibunda, Trachemys decussata* |
| Taxonomy | *Anolis distichus, Trachemys spp.* |
| Ecology | *Boiga irregularis* |

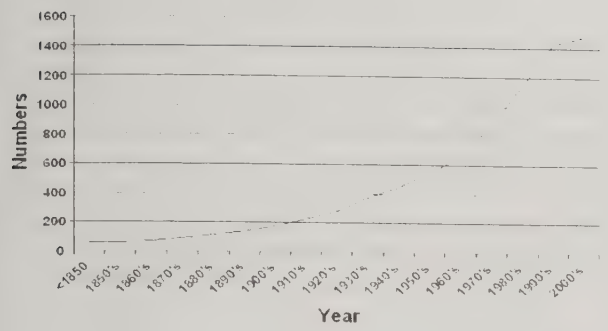



**Figure 1.** Cumulative growth in reptile and amphibian introductions from 1850–2005. Curve from 1850-1999 (before artifactitious decline resulting from reporting time lag) is approximated by the exponential growth equation $y = 46.014e^{0.2271x}$, with $R^2 = 0.9993$ and a doubling time of 30.4 years.

## HOW MIGHT ALIEN HERP INVASIONS BE FEASIBLY MANAGED?

Mitigation of impacts from invasive pests logically involves either preventing introductions in the first place, detecting and quickly eradicating new incursions, or managing entrenched populations so as to minimize their long-term impacts. I argue that the crypsis, high reproductive output, and high population densities that characterize many invasive herps make the latter two mitigation measures impossible to achieve in most circumstances. Very few examples of eradication or control have been attempted for alien herps. I will briefly review two examples that, I believe, are illustrative of the biological and social limitations that will frequently hinder such attempts.

Coqui frogs (*Eleutherodactylus coqui*) were introduced into Hawaii in the late 1980s or early 1990s in nursery plants imported from Puerto Rico. Immediately upon discovering this species in February, 1997, I repeatedly urged State officials having the responsible authorities to eradicate the species before it was irredeemably beyond control. This advice was motivated by knowledge of the high population densities the frogs achieve, recognition that their establishment would insert a novel trophic level into Hawaiian forest communities, and appreciation that their loud calls make the males readily detectable and targetable (as well as obnoxious), a necessary condition for any eradication campaign (Kraus et al. 1999). Despite this evidence and the damage that could reasonably be predicted to follow widespread establishment, State officials resisted efforts to intervene in this invasion for years, until well after the narrow window had passed during which statewide eradication was possible (Figure 2). The reasons for this failure resulted from the intersection of the species' high intrinsic rate of increase with the social constraints that (1) officials did not believe that a tiny frog could constitute a pest problem, and (2) effective control methods were lacking and took years to identify (Kraus and Campbell 2002). The biological parameter enforced a narrow window of control opportunity before populations were irredeemably entrenched; the social limitations emasculated response options until after that window had passed.

A somewhat similar dynamic pitting high intrinsic growth rates of an alien herp against lack of human imagination characterized the recent invasion of Burmese pythons (*Python molurus*) in southern Florida. Despite having scores of captures and reliable sightings of pythons in Everglades National Park by 2002 (including 27 in that year alone), biologists consulted to assess whether a population might be established advised that all represented released pets and posed no cause for concern (R. Snow, Everglades National Park, personal communication). It was clear to everyone even two years later (and clear in 2002 to anyone with invasive reptile experience) that pythons were widespread within the park and reproducing. They are now known to occur in high population densities and to range over an extensive area, including well beyond park boundaries (Snow et al. 2007). Again, the narrow window of opportunity for effective human response passed before it was even recognized.

Both these examples illustrate the common reaction from managers and many scientists to invasive herps: disbelief that a problem exists. And both species were allowed to expand beyond the point of effective control despite the fact that certain control advantages were present in these instances that would not obtain for most alien reptiles and amphibians. In particular, the problem of crypsis, which seriously hampers detecting most herp species, did not apply in the case of the coqui because the males' loud calls make them readily targetable for control. And the extremely large size of the pythons (>7 m), which certainly makes them more conspicuous than most alien herps, provided no advantage for early recognition of their naturalization. I suggest that both of these

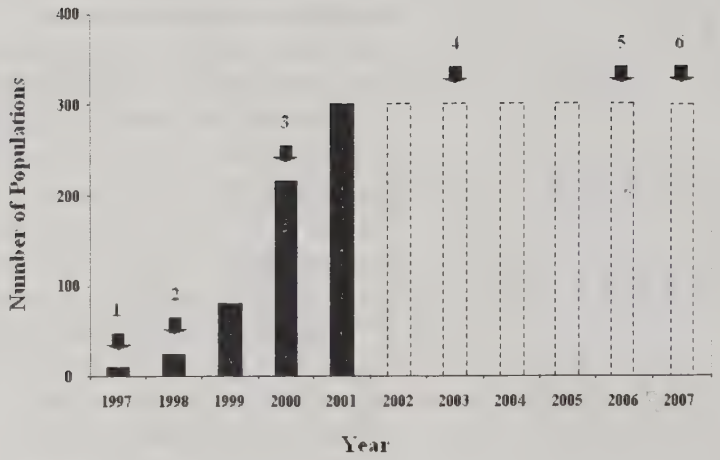



**Figure 2.** Contrast in growth in numbers of Hawaiian populations of coqui (*Eleutherodactylus coqui*) with milestones of State government response. Populations became too numerous to track after mid-2001 but presumably continued to grow exponentially, at least for a time. Despite that presumption, I merely outline the 2001 figures for all remaining years; estimated infestation by 2006 on Hawaii Island alone was >5000 acres (M. Wilkinson, HI Dept. Land & Nat. Res., personal communication). 1 = Alarm first raised by DLNR and USDA employees regarding threat posed by coqui, March 1997; 2 = first significant media coverage of coqui threat, November 1998; 3 = first State support to identify means of chemical control for coqui, June 2000; 4 = first Legislative action to facilitate control of coqui, indemnifying landowners to allow access for control operations, May 2003; 5 = first quarantine measures requiring treatment of inter-island shipments from infested nurseries, pending as of February 2006; Legislature declares coqui a pest, May 2006.

examples were unusual only in that biological weaknesses existed that could potentially have been exploited for effective control. However, these advantages were squandered by blinkered human responses that did not recognize the evidence of or admit the potential seriousness of the situation.

On the other hand, the coqui and python examples well exemplify the high reproductive rates (Kraus et al. 1999, Kraus and Cravalho 2001, Snow et al. 2007) that typify many other alien herp species. These high intrinsic population growth rates create very narrow windows of opportunity for implementing successful control. It is true that early detection and eradication of alien herps will sometimes be possible (cf., Whitaker and Bejakovich 2000 for Eastern banjo frog [*Limnodynastes dumerilii*] in New Zealand, and Fisher and Garner 2007 for bullfrogs in Britain). However, I suggest that most alien herp species will be even more difficult to control than coquis and pythons because (1) we will typically lack the benefit of having a readily identified biological fulcrum against which to act, (2) other species are likely to be more cryptic, and (3) psychological unresponsiveness to alien herps is likely to persist for the foreseeable future. From these considerations, I conclude that meaningful management of alien reptiles and amphibians must largely rely on prevention of their introduction. But design and implementation of effective prevention programs requires knowledge of how and why these animals are being transported. In short, one must understand the pathways by which alien reptiles and amphibians are moved and how those pathways vary.

## HOW ARE ALIEN REPTILES AND AMPHIBIANS BEING TRANSPORTED?

Only one study (Kraus 2003) has quantified how alien herps are being transported by humans and that study was preliminary to the expanded database analyzed in Kraus (in prep.) and briefly summarized here. Kraus (2003) showed, based on a dataset of 577 introduction records, that six pathways have accounted for most herp

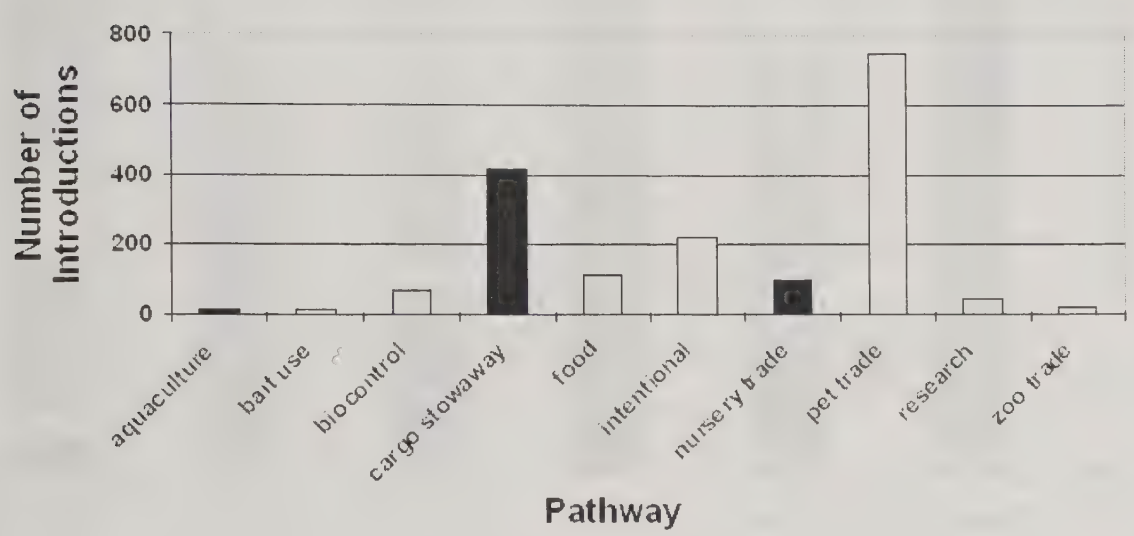



**Figure 3.** Numbers of introductions of alien reptiles and amphibians by pathway, 1850-2005. Open bars indicate intentional pathways; closed bars accidental pathways. "Intentional" refers to deliberate private introductions for aesthetic self-indulgence or other purpose.

introductions and that pathway importance varied taxonomically, temporally, and geographically. These major conclusions are upheld by the far more complete sampling (>5,700 introduction records) now available. Unlike most higher-level taxa, both intentional and accidental pathways have been important in the introduction of reptiles and amphibians (Figure 3). Intentional pathways include introductions for biocontrol, food use, the pet trade, and personal release of animals for aesthetic self-indulgence. Accidental pathways include transport as aquacultural contaminants, cargo stowaways, and hitch-hikers in nursery plant materials, the last a special subcategory of cargo stowaways deserving of separate tracking because of its increasing importance.

Frogs and lizards have been the most commonly transported herps and are the only major taxa to have utilized all six major pathways (Figure 4). Snakes and turtles have been introduced via a number of pathways too, but at lower levels, and salamanders and crocodilians have been transported relatively rarely (Figure 4). Only the pet trade and personal aesthetic pathways have involved all higher-level taxa, whereas the other four pathways have been more taxonomically limited in scope (Figure 4). Relative success in leading to naturalization varies among pathways, with transport via the nursery, biocontrol, and food pathways being far more likely to lead to successful establishment than transport via the other three (Kraus 2003, in prep.).

Pathway importance has varied temporally, with the cargo pathway being of greatest importance for the second half of the 19th Century, intentional aesthetic introductions of greatest importance for the first part of the 20th Century, and the pet trade of greatest predominance since the 1970s (Figure 5). The biocontrol pathway has been largely quiescent for the past several decades, the food-use pathway has grown relatively little during that same

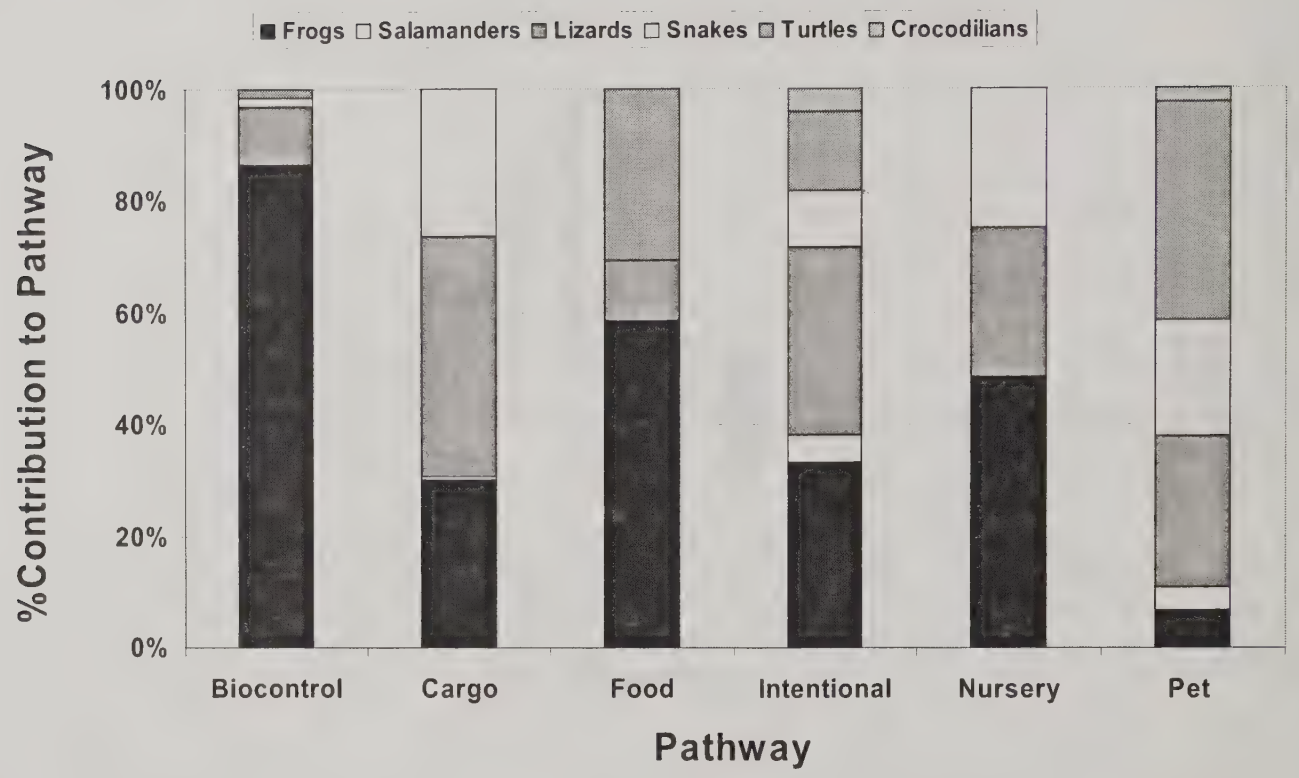



**Figure 4.** Relative taxonomic distribution among the six primary introduction pathways, 1850-2005. "Intentional" refers to deliberate private introductions for aesthetic self-indulgence.

period, but the nursery-trade pathway has increased considerably in importance since the 1970s (Figure 5). Introduction rates from 1980-2005 have been considerably higher for the pet-trade, nursery-trade, and cargo pathways than for the period 1850-1979 (Kraus, in prep.). Rates of introduction for biocontrol have been drastically less during that same period, while rates for food use and intentional aesthetic introductions have remained unchanged (Kraus, in prep.).

The large majority of documented introductions have involved Europe and North America (Kraus, in prep.). Kraus (2003) showed pathway importance varies among the US, Caribbean, and Pacific regions, and this pattern holds true for other regions as well. Generally speaking, the pet trade is the most important pathway for introductions into North America, Europe, and Asia, whereas cargo has predominated in the Pacific, and cargo and the nursery plant trade are co-dominant in the Caribbean (Figure 6). The upshot is that each major geographical region has its own unique signature of pathway representation and relative importance (Kraus, in prep.).

## HOW MIGHT EFFECTIVE PREVENTION PROGRAMS BE DESIGNED FOR ALIEN HERPS?

The information above has important ramifications for designing and implementing effective prevention strategies for invasive reptiles and amphibians. First, strategies to abate intentional versus accidental introductions are quite different, but both will need to be included to limit further alien herp incursions. Intentional introductions may be addressed by implementing some form of taxon-based risk-assessment system incorporating features that appear diagnostic for either naturalization or invasiveness, as is done for weeds (Pheloung et al. 1999, Walton et al. 1999, Williams et al. 2002) and some vertebrates (Bomford and Hart 1998, Kolar and Lodge 2002, Bomford 2003, Bomford and Glover 2004). Such a method requires identification of ecological or other predictors of naturalization/invasiveness for

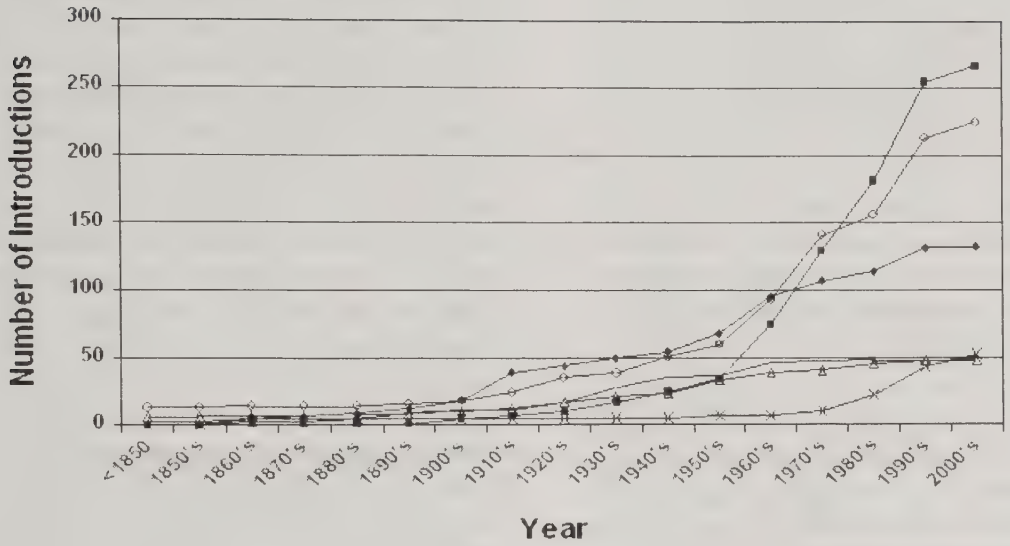



**Figure 5.** Changes in global pathway importance for the six major pathways, 1850-2005. Solid line, biocontrol; open circles, cargo; open triangles, food use; X's, nursery trade; filled squares, pet trade; filled diamonds, intentional aesthetic releases.

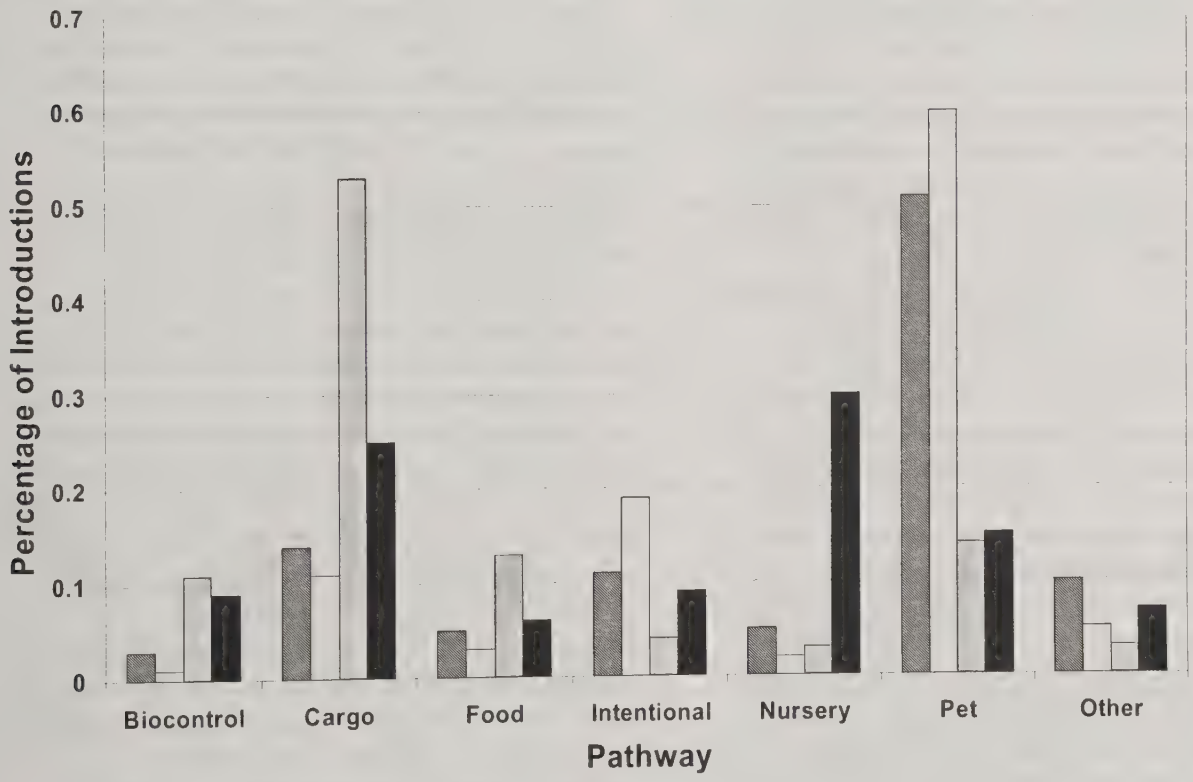



**Figure 6.** Variation in pathway importance by geographic region for all documented introductions, 1850-2005. Diagonal lines, North America; stippling, Europe; horizontal lines, Pacific region; and solid bars, Caribbean.

the targeted taxa. Higher numbers of predictive variables should be expected to lead to more robust results, as typifies the weed-risk-assessment systems (Pheloung et al. 1999, Daehler et al. 2004).

Bomford et al. (2005) tested the utility of four variables for their usefulness in predicting naturalization among alien reptiles and amphibians introduced to Great Britain, California, and Florida. They found that degree of climate match between native and introduced ranges, a taxonomic-risk variable, and whether a species had previously established elsewhere were useful in segregating among species that successfully established in these three jurisdictions versus those that did not. In contrast, they found that size of native range (thought to reflect breadth of ecological tolerance) had no predictive value. This study is currently being refined, and a larger number of variables needs to be included, but their results suggest that such an approach may be promising for predicting naturalization ability among reptiles and amphibians in particular regions, thereby providing a tool for screening intentionally introduced species. Efforts will also soon be made to identify additional ecological variables that may refine this approach.

Approaches toward preventing reptile and amphibian introductions via accidental pathways (primarily cargo and nursery plant trade in my study, but also including lesser numbers introduced on vehicles and as aquacultural contaminants) are more tentative. Recall that each geographical region exhibited its own characteristic signature of pathway representation and importance. It may prove possible to correlate these patterns with simple economic or legal variables for any country within each region and use those indicators to predict which pathways merit greatest attention within any particular jurisdiction. Should that prove true, one could then prioritize limited resources to those pathways so identified, thereby maximizing prevention efficiency. For example, should magnitude of nursery-trade introductions correlate with income levels, assessing the magnitude of the latter could be used to predict the risk of the former and inform decisions as to how much effort to devote to searching or treating that pathway. Assessing whether economic indicators correlate with pathway intensity is the subject of ongoing investigation by the author along with B. Kaiser and K. Burnett.

Should such economic predictors be identified, it would still leave unresolved how animals in the accidental pathways are best intercepted. All such pathways must involve some form of port-of-entry inspection program, relying on the border quarantine programs widely in place around the world. But such programs necessarily sample only a very small portion of arriving goods and must rely on repeated sampling schemes to identify highest-risk pathways and provide quality assurance. This is where prioritization of inspection effort informed by economic predictiveness of pathway importance may prove useful. However, treatment methods to remove hitch-hiking herps, other than hand-capture, have been little investigated and we require development of methods of scale that can effectively treat entire cargo shipments. As one example of what is needed in this regard, a short hot-water drench is sufficient to kill hitch-hiking frogs in potted nursery plants (F. Kraus unpublished data), a pathway of increasing importance, but is just beginning to be scaled up to a level commensurate with industry volume in Hawaii (D. Cravalho, Hawaii Dept. Agriculture, personal communication). Similar scaled-up methods that go beyond visual inspection and hand-capture are needed to more effectively address reptile transport via this same pathway and to address all species transported in other accidental pathways.

Note that both prevention options discussed above – taxonomic risk-assessment systems for intentional introductions and identification of economic indicators to assess risk of accidental pathways – serve (ideally) only to best identify those taxa or pathways of highest risk for inadvertent importation or establishment. In many countries, having that information does not necessarily mean it will be used for interdiction, due to a variety of political, legal, and procedural hurdles discussed in detail by Simberloff (2005) for the case of the US. For intentional introductions, use is required of a comprehensive white list/black list system that explicitly adopts the precautionary principle and forbids importation until a species has been assessed as posing little risk. Such methods may have their statistical limitations (cf. Smith et al. 1999, Caley et al. 2006) but are vast improvements over the minimal, reactive programs that typify the majority of countries. In the case of the US specifically, this reactive stance is embodied in the Lacey Act, part of which includes a miniscule list of "injurious wildlife" banned from import because they have proven harmful elsewhere. A problem with that reactive approach, of course, is that only a tiny fraction of proven invasive species – those that have garnered some

level of political concern – are included on that list and high-risk species that happen not to have been imported anywhere yet are ignored entirely. Similar problems attend assessments of risk for particular pathways, which typically assess only a limited pool of likely risks and impacts (Simberloff 2005). It is worth noting that continued pursuit of such minimalist policies for preventing invasions is inconsistent with the risk-analyses outlined here, which are intended to identify high-risk taxa and pathways so that exclusionary measures might actually be implemented. As far as I am aware, a comprehensive approach based on the precautionary principle seems to have been adopted to date only by Australia and New Zealand.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

Bomford, M. 2003. Risk assessment for the import and keeping of exotic vertebrates in Australia. Bureau of Rural Sciences, Canberra, Australia.

Bomford, M., and J. Glover. 2004. Risk assessment model for the import and keeping of exotic freshwater and estuarine finfish. Bureau of Rural Sciences, Canberra, Australia.

Bomford, M., and Q. Hart. 1998. Risk assessment for importing and keeping exotic vertebrates. Pages 406-410 *in* R. O. Baker and A. C. Crabb, editors. Proceedings of the 18th Vertebrate Pest Conference, University of California, Davis, California, USA.

Bomford, M., F. Kraus, M. Braysher, L. Walter, and L. Brown. 2005. Risk assessment model for the import and keeping of exotic reptiles and amphibians. Final report to Australian Department of Environment and Heritage.

Caley, P., W. M. Lonsdale, and P. C. Pheloung. 2006. Quantifying uncertainty in predictions of invasiveness, with emphasis on weed risk assessment. Biological Invasions 8:1595-1604.

Capula, M., L. Luiselli, M. A. Bologna, and A. Ceccarelli. 2002. The decline of the Aeolian wall lizard, *Podarcis raffonei*: causes and conservation proposals. Oryx 36:66-72.

Daehler, C. C., J. S. Denslow, S. Ansari, and H. C. Kuo. 2004. A risk-assessment system for screening out invasive pest plants from Hawaii and other Pacific islands. Conservation Biology 18:360-368.

Fisher, M. C., and T. W. J. Garner. 2007. The relationship between the emergence of *Batrachochytrium dendrobatidis*, the international trade in amphibians and introduced amphibian species. Fungal Biology Reviews 21:2-9.

Kolar, C. S., and D. M. Lodge. 2002. Ecological predictions and risk assessment for alien fishes in North America. Science 298:1233-1236.

Kraus, F. 2003. Invasion pathways for terrestrial vertebrates. Pages 68-92 *in* J. Carlton, G. Ruiz, and R. Mack, editors. Invasive species: vectors and management strategies. Island Press, Washington, D.C., USA.

Kraus, F., and E. Campbell. 2002. Human-mediated escalation of a formerly eradicable problem: the invasion of Caribbean frogs in the Hawaiian Islands. Biological Invasions 4:327-332.

Kraus, F., and D. Cravalho. 2001. The risk to Hawaii from snakes. Pacific Science 55:409-417.

Kraus, F., E. W. Campbell, A. Allison, and T. Pratt. 1999. *Eleutherodactylus* frog introductions to Hawaii. Herpetological Review 30:21-25.

Pheloung, P. C., P. A. Williams, and S. R. Halloy. 1999. A weed risk assessment model for use as a biosecurity tool evaluating plant introductions. Journal of Environmental Management 57:239-251.

Simberloff, D. 2005. The politics of assessing risk for biological invasions: the USA as a case study. Trends in Ecology and Evolution 20:216-222.

Smith, C. S., W. M. Lonsdale, and J. Fortune. 1999. When to ignore advice: invasion predictions and decision theory. Biological Invasions 1:89-96.

Snow, R. W., K. L. Krysko, K. M. Enge, L. Oberhofer, A. Warren-Bradley, and L. Wilkins. 2007. Introduced populations of *Boa constrictor* (Boidae) and *Python molurus bivittatus* (Pythonidae) in southern Florida. Pages 416-438 *in* R. W. Henderson and R. Powell, editors. Biology of the boas and pythons. Eagle Mountain Publishing, Eagle Mountain, Utah, USA.

Walton, C., N. Ellis, and P. Pheloung. 1999. A manual for using the weed risk assessment system (WRA) to assess new plants. Australian Quarantine and Inspection Service, Canberra, Australia.

Whitaker, T., and D. Bejakovich. 2000. Exotic frog incursion. Surveillance 27:12-14.

Williams, P. A., A. Wilton, and N. Spencer. 2002. A proposed conservation weed risk assessment system for the New Zealand border. Science for Conservation 208:1- 47.

# ISLAND BIOSECURITY AS A PEST MANAGEMENT TACTIC IN NEW ZEALAND

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***Abstract:*** New Zealand is an archipelago with many islands of conservation significance, none of which has the full suite of invasive vertebrate pests found on mainland New Zealand. Managing invasive species on New Zealand islands involves prevention of pests establishing and controlling or extirpating those already there. Prevention measures, referred to as island biosecurity, focuses on the three major pathways for pests to arrive: deliberate release, accidental release and swimming. Managing deliberate release is largely reactive but does have opportunities for prevention depending on the motivation of offenders. Managing accidental release is where most proactive work is done by Department of Conservation staff to protect remote high-value nature reserve islands. A comprehensive quarantine, surveillance and contingency response system is in place for these sites. When managing self introductions, the size and nature of the water gap and the species involved largely determine the risk. Many islands are beyond the swimming range of all pests, but for some closer ones, we are exploring the feasibility, costs and benefits of managing reinvasion across various water gaps and trying to improve our detection and elimination techniques. Eradication backed up by successful ongoing island biosecurity in many cases compares well with alternative management options at mainland sites.

***Key Words:*** biosecurity system, incursion response, invasive species, islands, pest management, prevention, surveillance.

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## INTRODUCTION

New Zealand is an archipelago of more than 2000 islands (greater than 0.1 ha) covering 23 degrees of latitude. Many of these islands have some conservation significance. Mainland New Zealand (the two largest islands) has 28 introduced mammal and many other exotic fish, bird and herpetofauna species. None of the other islands has this full suite of invasive vertebrate pests. Therefore, managing invasive species on New Zealand islands gives priority to prevention as well as taking opportunities toward restoration through pest eradication. The costs and risks of managing reinvasion is a pre-requisite consideration to removing pests from islands.

Preventing pests from establishing a breeding population on islands is the goal of island biosecurity, a term used in New Zealand to encompass quarantine (prevention), surveillance (detection), and response to incursions (contingency). To achieve this goal, the focus is on managing pathways for island invasion. Invasive vertebrate species reach islands through either deliberate or accidental release by people, or they swim (self introductions) from nearby islands or the mainland islands.

This paper describes actions taken by New Zealand Department of Conservation (DOC) to address each pathway.

## DELIBERATE RELEASE

People release animals on islands deliberately for three reasons:

- They didn't think or know about the potential consequences
- They want to take advantage of commercial or recreational opportunities
- They acted maliciously because of anti-conservation or anti-government sentiments

### Didn't Think - Didn't Know

The most common scenario involves visitors bringing pets (e.g., ferrets [*Mustela furo*] or brushtail possums [*Trichosurus vulpecula*]) to islands and allowing them to escape. Pet owners are usually unaware that the island does not have these species in the wild and are reluctant to accept that their pet could do any harm. From their perspective, they are "letting them go free to live out their life in the wild".

**Commercial or Recreational Opportunity**

Releases in this category are usually for hunting opportunities, either for those people involved or for fee-paying clients. Commercially-motivated releases are rare today, but prevalence is connected to the monetary value of the animals or the enterprise. Those people engaged in subsistence living "off the land" often regard the release of animals as within their rights. Examples of species released for this reason include deer (*Cervus* spp.), pigs (*Sus scrofa*), and in the past, possums.

**Malicious Release**

Such releases most often are not actually carried out, but are threatened or set up as a hoax. They usually are precipitated by grievances among participants. In some cases, the offender has a completely different world view of introduced species. For example, an Auckland man deliberately set out to establish a wild population of rainbow lorikeets (*Trichoglossus haematodus*) because he felt they would "add colour to the local wildlife".

There are three ways of managing the deliberate release pathway: information, law enforcement, and contingency response.

**Information**

Increasing public awareness of the potential threats and consequences of deliberate release is an important component of the overall strategy. This can be done in different ways, including distributing pamphlets and permits to island visitors, engaging media coverage of incursion responses or prosecution, and consulting with people affected by conservation-related activities.

**Law Enforcement**

New Zealand law is relatively stringent in these matters, but bringing perpetrators to justice is always challenging. For example, the maximum penalty for deliberately releasing a wild animal without authority is NZ$50,000. Under other legislation, penalties for similar offences range up to $100,000. However, maximum penalties are rarely, if ever, handed down.

**Contingency Response**

Responding to information received on deliberate releases is vital to demonstrate commitment to our own prevention messages. Failure to respond professionally undermines credibility. DOC has a process which facilitates information collation and decision-making on the most appropriate response. We follow this process even when deciding not to take further action.

## ACCIDENTAL RELEASE

Stowaway pest animals potentially impact most on nature reserve islands. These islands have restricted public access and are, therefore, less vulnerable to deliberate release. DOC has the greatest opportunity to manage this pathway, especially on reserve islands where most visitation is by DOC employees or scientists operating under DOC supervision. DOC's island biosecurity system aims to prevent pests reaching islands accidentally with the supplies and equipment of people visiting. For islands outside DOC control, DOC raises awareness of the accidental release pathway and advocates others to follow DOC's example.

## SWIMMERS

The size and nature of the water gap and the invasive species involved appear to be most important predictors of invasion risk through this pathway. For islands far offshore, pest animals unable to fly have virtually no chance of arriving unassisted. However, for islands closer to the mainland DOC is exploring the feasibility, costs and benefits of managing reinvasion across water gaps of various sizes.

For example, stoats (*Mustela erminea*) are accomplished swimmers that can cross water gaps up to 3 km to reach islands. The eradication of stoats from several islands in Fiordland and subsequent modelling of their reinvasion indicates that islands beyond 500 m have a manageably low reinvasion rate by swimming stoats – about once per 3 years.

Similar work is underway for rodents in an effort to understand the frequency of invasions by swimming and also to improve the technology for efficient surveillance and contingency response to better manage reinvasion. For example, Norway rats (*Rattus norvegicus*) have been detected arriving about once per year on average over the last 10 years on Ulva Island from Stewart Island, a distance of about 800 m. Norway rats equipped with radio transmitters and released on rat-free islands are variable in their behaviour, but tend to go through an exploratory phase for about two weeks before settling into a territory.

Managing islands within the swimming range of pest animals is less efficient (because of reinvasion risk) than more remote islands, but may offer a

better alternative than managing pests on the mainland where reinvasion is constant.

## THE DOC ISLAND BIOSECURITY SYSTEM

The first opportunity to prevent pest animals from establishing on islands through accidental release by DOC staff is at the source of expedition supplies. For example, fresh vegetables are no longer bought from home gardens or roadside vendors; they are sourced from supermarkets with good quality control systems so that their suppliers eliminate animal infestation. The second opportunity comes from checking provisions and equipment through a quarantine store and packing everything in suitable containers to deny entry by pests after checking.

Further opportunities arise at departure points, in transit, and in some cases, upon arrival where DOC rangers require unpacking and checking of equipment in a sealed room for a final time. Ongoing surveillance and a contingency incursion response capability for pests considered most likely to arrive (e.g., rodents) are also required to cover unauthorised landings and shipwrecks.

As each opportunity to detect and eliminate pest animals passes, the risk of pests establishing increases for many organisms. By the time pests have actually arrived on the island, the chances of finding and eradicating them before they reproduce is much reduced (Figure 1).

At the national level, key components of the DOC system are as follows:

- A Standard Operating Procedure (SOP) that provides the overall structure and standards
- A Best Practices Manual that provides a basis for sharing current knowledge on the tools and methods available to meet island biosecurity goals and standards
- An incidents database on pest incursions and interceptions that can be used to improve the system and to demonstrate to people that incursions really do happen

The SOP and Best Practices Manual provide the basis for writing island biosecurity plans for each DOC conservancy. These plans set the biosecurity standards expected of people visiting islands. The system is audited to look at actual biosecurity practices in the field compared with standards laid down in the plan. Attempts are made to resolve differences through modifying field practice or plan standards. Audits also identify new best practices that can be shared with other conservancies (Figure 2).

Conservancy plans provide an incursion response procedure to allow information to be gathered and appropriate action to be planned with help from the Best Practices Manual. Every incident, be it an incursion or a near miss, is logged into the database, and there is some investigation into how well the system worked and what could be improved for the future.

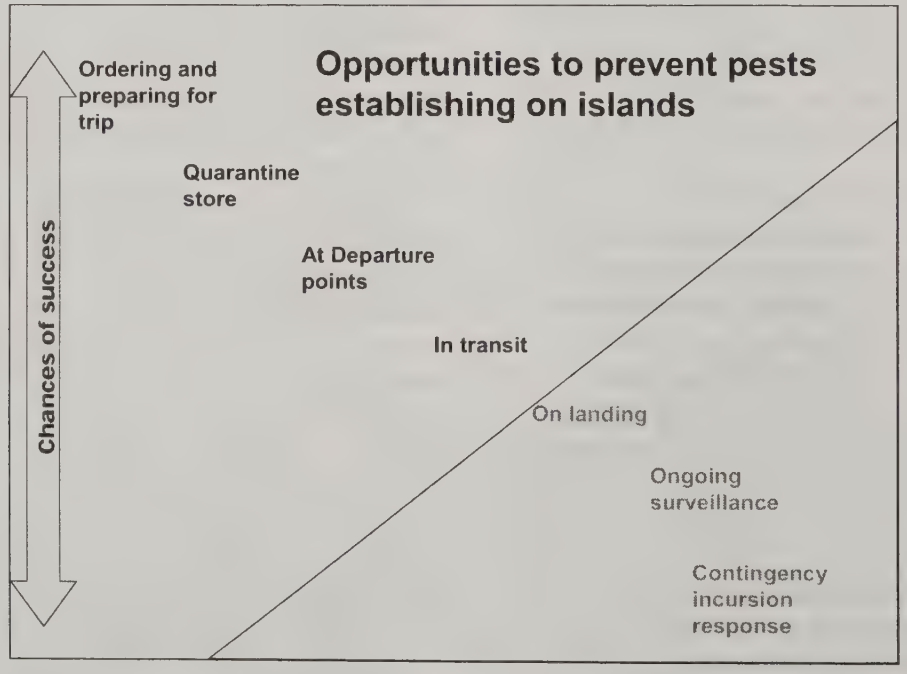



**Figure 1.** Intervention points to prevent invasive pest animals from establishing populations on islands.

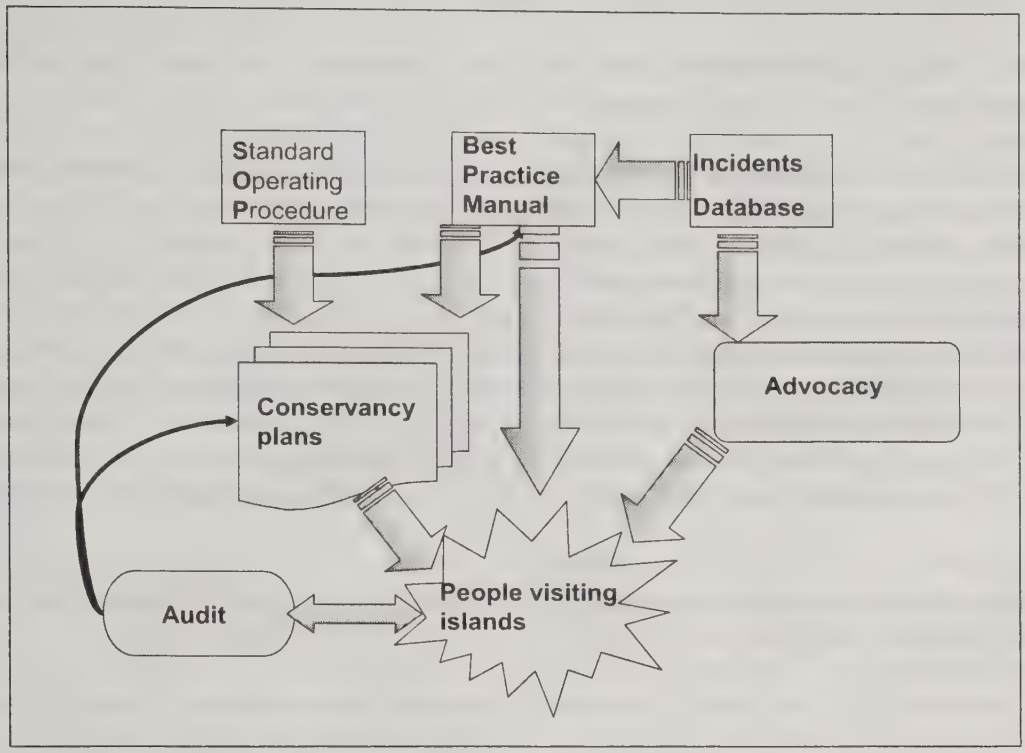



**Figure 2.** The New Zealand Department of Conservation's island biosecurity system.

## CONCLUSION

The invasive pest animal status of our islands is changing, and there is potential for things to get worse through inaction. Island biosecurity is an important part of New Zealand's overall management of invasive pests, not only in prevention of further invasions, but also securing gains made in restoration (eradication) projects and as a cost-effective alternative to mainland pest management. The key to success is being effective across as many pathways as possible. The opportunity for further benefits lies in increased effort to raise public awareness and to support island communities taking biosecurity actions for themselves.

# INVASIVE SPECIES MANAGEMENT AND RESEARCH USING GIS

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***Abstract:*** Geographical Information Systems (GIS) are powerful tools in the field of invasive species management. GIS can be used to create potential distribution maps for all manner of taxa, including plants, animals, and diseases. GIS also performs well in the early detection and rapid assessment of invasive species. Here, we used GIS applications to investigate species richness and invasion patterns in fish in the United States (US) at the 6-digit Hydrologic Unit Code (HUC) level. We also created maps of potential spread of the cane toad (*Bufo marinus*) in the southeastern US at the 8-digit HUC level using regression and environmental envelope techniques. Equipped with this potential map, resource managers can target their field surveys to areas most vulnerable to invasion. Advances in GIS technology, maps, data, and many of these techniques can be found on websites such as the National Institute of Invasive Species Science (www.NIISS.org). Such websites provide a forum for data sharing and analysis that is an invaluable service to the invasive species community.

***Key Words:*** buffer, early detection, environmental envelope, geographic information systems, GIS, invasive species, regression, thiessen polygons.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W.C. Pitt, K.A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Geographic Information Systems (GIS) and Global Positioning Systems (GPS) provide a mechanism to digitally pinpoint a location on earth, view the location on a map, and use the location and ancillary data in spatial analyses. Individuals are able to quickly and easily produce maps and conduct spatial analyses that would otherwise be difficult or possible to produce using a network of satellites, satellite receivers, and mapping software. GIS serves as a data storage and analysis device for spatial data, making data easy to view and manipulate.

Health care, agriculture, and environmental industries are a few of many entities that have been positively influenced with the advent of GIS technologies. Large spatial databases can help companies track their hard goods and allow farmers to determine which areas of their fields need more fertilizer, eliminating the need to add fertilizer to the entire field. Ecological data often contains spatial component. Where an animal spends its time and the patterns of its movements can be important clues to its biology. Biological information of non-native species provides insight to explaining expanding distributions and provides a watch list of spreading invasive species to managers for early detection and rapid response.

GIS can be a useful tool for monitoring invasive vertebrates, especially for early detection and rapid assessment. Species distributions are largely determined by the environment. A growing number of statistical models, called Species Environmental Matching (SEM) models are being used to determine current and potential distributions and abundances of harmful invasive species (Stohlgren and Schnase 2006). SEM models relate observed species distributions to environmental (climatic, topographic, edaphic) envelopes. Then, assuming the same stable relationships, they project species spatial shifts (local, enrichment, or extinction) in response to envelope changes under current conditions. These environmental envelopes, arranged along a gradient from proximal to distal predictors, may have direct or indirect effects on species' establishment and survival (Austin 2002). SEM models are either created in a GIS or can be displayed in GIS to give a visual representation of the environmental envelope and potential habitat or abundance.

An important consideration for invasive species management is that recent invaders may not have filled all suitable habitats, while species naturalized long ago may have filled a larger proportion of suitable habitat. Defining where a species may survive depends heavily on being able to determine

its existing or potential habitat. Technological advances in GIS software make these types of analyses more readily accessible to the general public. Data such as elevation, vegetation type, and climate information are now available for free on the internet, often paired with websites that allow persons to view and utilize the information.

## HOW GIS CAN BE USED

### View Data

A very basic and effective way to use GIS is to view data. Many datasets are very large and difficult to visualize as a table of numbers. When viewed spatially, these data often make more sense. Stohlgren et al. (2006) combined native and non-native fish datasets from NatureServe and the United States Geological Survey (USGS) Florida Integrated Science Center's Non-Indigenous Aquatic Species database to examine numbers of native and non-native species in each 6-digit HUC area (Figure 1). Without performing any statistical analyses, they found a large majority of native fish in the US are centered throughout the mid-west and south-central US. The non-native fish are found primarily in the western and eastern US. These patterns were ascertained without conducting any statistical analysis, proving that displaying the data in a spatial format can be a useful endeavor even without detailed analyses.

### Data Summary

GIS can be helpful for summarizing large datasets for modeling habitat quality and distribution. Data layers, such as Digital Elevation Models (DEM), are often used in modeling because they provide a large amount of environmental information. DEMs are available free via the internet, often at either 10- or 30-meter resolutions. While these resolutions provide a lot of information at a fine scale, this scale is not always necessary or desired. In certain situations, GIS can be used to summarize data, often simplifying the data into the resolution of interest. The Spatial Analyst module of ArcGIS 9 (ESRI 2004) has a zonal statistics function that calculates raster layer summary statistics for a large polygonal area like a county. This module will extract the average, minimum or maximum, and range for each polygon. Additionally, GIS can be used to extract the value for a specific point in a DEM so that the entire surface does not have to be stored. GIS functions like these make the retrieval of dependant data for models readily accessible.

### Field Data - Points, Lines, and Polygons

Spatial field data can be displayed and managed in a GIS. The data are stored in one of three formats: points, lines, or polygons. Locations of individual organisms are examples of points. These are discrete one-dimensional places in space. A linear representation of interest is a line. Lines include rivers, transects, or roads. Polygons represent an area of interest, like a stand of trees or a lake.

Point data types are an excellent medium for recording presence or absence of a species because they are discreet. Lines give similar amounts of information, again lending themselves well to presence, absence, and additional attributes. Polygons are unique because they cover an area which can contain additional information such as abundance or percent cover. All these data types can be collected in the field using either paper maps or GPS devices to collect the data before downloading it to a computer.

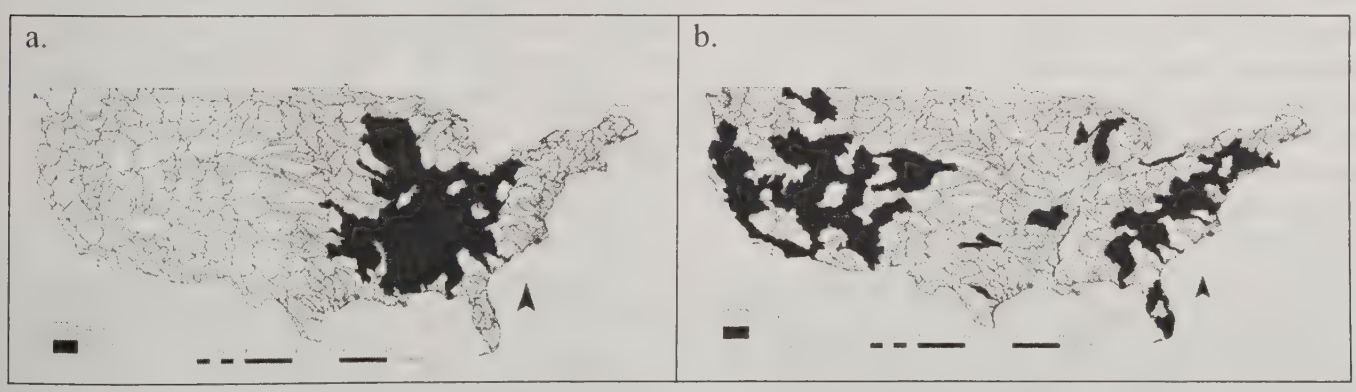



**Figure 1.** Pattterns of (a.) native and (b.) non-native fish by 6-digit HUC drainage (Stohlgren et al. 2006)

**Simple GIS models**

GIS can be used to create simple analyses such as buffers and thiessen polygons. A buffer can be created around points, lines, or polygons. It is a new polygon of specified distance from the original feature. Any GIS program can create a buffer around points, lines, or polygons that can be used for various reasons, such as surrogates for habitat for poorly studied species. Buffers can also define potential habitat for species that have a very specific distance they can be from a given feature, such as water. Buffers are a commonly used transformation of spatial data.

Another analysis performed by GIS is the creation of thiessen polygons, sometimes known as voronoi polygons. Theissen polygons are created around a group of points, one polygon for each point. The polygons are created around each point in such a way that every location lies within the polygon of the point to which it is nearest. The easiest way to think about thiessen polygons is with fast food delivery areas. A fast food pizza chain would divide a city into thiessen polygons, only delivering to customers that were closer to them than they were to the next restaurant. A wildlife example would be, if there are twelve nests in an area, polygons are formed around those twelve nests so that every place on the landscape falls into the polygon associated with the closest nest. This tool has many applications for studying territorial animals. Nest location data could be used to generate thiessen polygons surrounding each nest. The area surrounding each nest could be an estimate of territory range. Buffers and thiessen polygons are two of the many possible examples of simple operations that can be done using a GIS.

**Statistical models**

Statistical models use current species distribution data to try and predict potential habitat. Conceptually, the SEM models assume the fitted observational relationships to be an adequate representation of the realized niche of a species under a stable equilibrium or quasi-equilibrium constraint. As such, the SEM model result is only a first approximation of future distributions of individual species (Pearson and Dawson 2003). SEM model results are also determined by other processes such as dispersal, adaptation, competition, succession, fire and grazing pressure (Austin 2002). Still, an integrated model may contribute considerably to a robust early warning system for decision makers to design more effective management and control strategies for harmful invasive species. In short, we will be able to better manage and assess risks associated with harmful invasive species because risk assessments require accurate modeling of current and potential species distributions (Stohlgren and Schnase 2006).

Numerous challenges exist in traditional SEM or niche-based modeling for current and future species distributions (see reviews by Pearson and Dawson 2003, Soberon and Peterson 2005, Elith et al. 2006, Guisan et al. 2006, Heikkinen et al. 2006, Hijmans and Graham 2006, Peterson 2006, Beaumont et al. 2007). These challenges have not prevented scientists and resource managers from refining, testing, and using SEM models in their work. No two SEM models are identical, and each has advantages and disadvantages (Table 1).

**Regression Models**

Logistic regression is a type of Generalized Linear Model (GLM) appropriate for data with a binary distribution such as species presence or absence (McCullagh and Nelder 1989). The output from logistic regression models can be taken from statistical software and used in GIS to create a visual representation of the model created. We have done this with data obtained from the USGS Florida Integrated Science Center's Non-Indigenous Aquatic Species database on the invasive cane toad (*Bufo marinus*). The cane toad has become established in the US and invaded several watersheds in Florida. We employed logistic regression with Systat 11.0 (SSI 2004) using minimum temperature, minimum radiation, mean temperature, maximum temperature, maximum humidity, and maximum growing degree days as predictor variables to determine how much potential habitat exists for the cane toad in the south-eastern US. We constructed a step-wise GLM, and only minimum temperature was selected as a significant variable. Results of the regression analysis had a high predictive power (McFadden's Rho Squared = 0.92). When the results were imputed in a GIS, the map showed that the cane toad had invaded most of its suitable habitat in the Florida area, with only a few un-invaded areas left in high and medium habitat suitability areas (Figure 2). This result was a first approximation model. More data and ecological information on the cane toad could produce better results in the future.

**The Environmental Envelope Model**

The Environmental Envelope Model (EEM, Jarnevich et al. 2007) was developed as a rapid assessment technique to estimate the potential

**Table 1.** Commonly used species environmental matching models for predicting species distributions.

| Model | Citation | Advantages | Disadvantages |
|---|---|---|---|
| Maxent | (Phillips et al. 2006) | Presence only, nonlinear, nonparametric, not sensitive to multicollinearity, provides variables' relative importance (jackknifing), easy to run and takes less time, becoming popular | Presence only (no consideration of absence data) |
| Classification and Regression Tree (CART) | (Breiman et al. 1984) | Non-parametric, Presence/absence, easy to run and interpret | Absence data needed |
| Boosted Regression Tree | (Friedman 2001, De'ath 2007) | Non-parametric, Presence/absence, limitations with spatial data | Absence data needed, more statistical details |
| Logistic Regression | (McCullagh and Nelder 1989) | Widely used, presence/absence | Absence data needed, sensitive to multicollinearity |
| Least square regression | Most statistics software | Widely used, continuous response variable (e.g., species richness) | Needs continuous response variable, sensitive to multicollinearity, decision about significance level (*P* value?) |
| BIOCLIM | (Busby 1991) | Presence only, simple | Presence only, does not use absences, less accurate than other niche models |
| DOMAIN | (Carpenter et al. 1993) | Presence only, simple | Presence only, does not use absences, less accurate than other niche models |
| ENFA (Env. Niche Factor Analysis) | (Hirzel et al. 2002) | Presence only | Presence only, does not use absences |
| Envelope | (Jarnevich et al. 2007) | Presence only or absence only models can be run. | All environmental factors are given equal weighting. |

distribution of a species given its present location and associated environmental attributes. It is supported by ArcGIS 9x (ESRI 2004) and will be available on the National Institute of Invasive Species Science (NIISS) website (www.NIISS.org). Envelope models use environmental variables, chosen by the modeler that are relevant to the species of interest or species growth in general, to determine locations within the environmental envelope where the species of interest may be able to become established. The minimum and maximum of each independent variable are noted

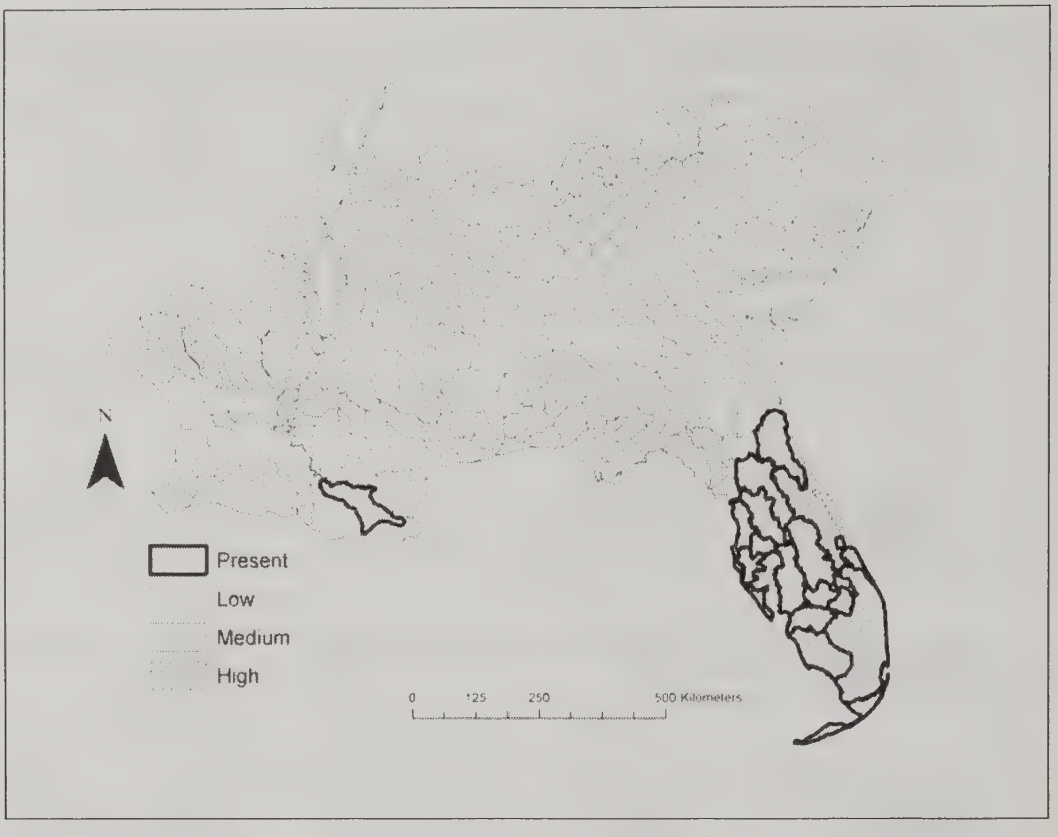



**Figure 2.** Regression model of the cane toad showing low, medium, and high likelihood of suitable habitat in each 6-digit HUC.

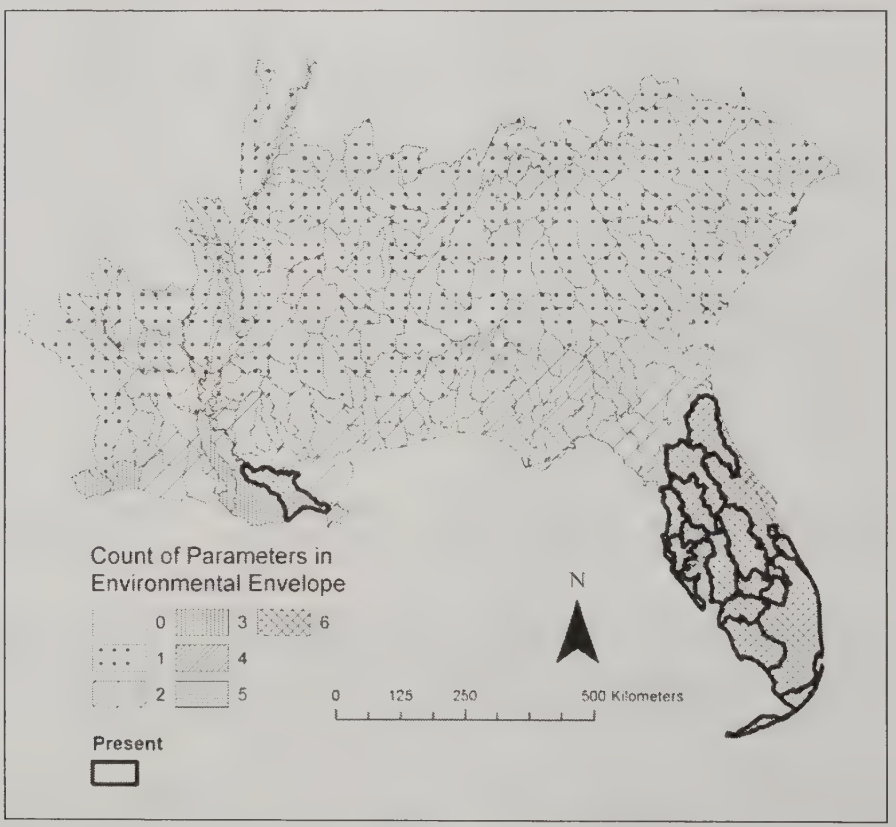



**Figure 3**. Envelope model of the cane toad showing the number of parameters in each 6-digit HUC that could contain the species.

by the ArcGIS program for all of the locations that the species is present. These minimum and maximum values together become the "envelope" in which the species can survive. For instance, if a species exists in only three counties and the temperature in county A is 45° F, county B is 40° F, and county C is 43° F, then the temperature envelope is 40 to 45° F. We would then compare the temperature for other counties to see if they fell within the range of potential habitats. The model can include several different environmental layers to determine suitable habitats. The output of the model informs how many of the input variables lie within the environmental envelope of the species.

We conducted an EEM analysis on the cane toad to compare results generated from the regression model (Figure 3). We used environmental data retrieved from the Daymet website (www.daymet.org) that was originally at 1 $km^2$ resolution for the dependant variables. We used zonal statistics in ArcGIS's Spatial Analyst (ESRI 2004) to summarize the data for the 6-digit HUCs. Variables included minimum radiation, minimum temperature, mean temperature, maximum temperature, maximum humidity, and growing degree days. We used the same cane toad data from the regression model. The resulting map showed that as distance increases from the peninsula of Florida, there are less environmental variables that fall within the cane toad's environmental envelope. This trend supports the regression model that showed the cane toad did not have much more suitable habitat than what is already occupied.

### GIS on the Web

Common issues confronting GIS users today include software and data availability and user friendliness. GIS software is often expensive, making it difficult for many people to obtain. Another subset of would-be GIS users have access to software, but do not have the time required to learn to effectively and efficiently use the software. These issues are changing with the advances in GIS technology. Many of the functions that are found in proprietary software can also be found on the internet. Much of the species distribution data used in the examples in this paper were found and downloaded from the internet. Many websites, such as NIISS are encouraging an environment of data sharing. The NIISS website includes an interface to upload data and a GIS interface to view data graphically, create models, and print and save final map products. The technology is very sophisticated and is open to the general public. This may be the direction GIS software is heading toward, reducing dependence on desktop GIS software in the future.

## CONCLUSION

Advances in GIS technology have made it become a useful tool for land managers and academics alike. It is widely used as a tool to perform basic functions such as displaying data and more complex functions like creating and displaying SEM models. As we look to our computers today and continue to look to the future of GIS technologies, GIS is a tool that should, and could be used by many scientists and resource managers alike.

## LITERATURE CITED


AUSTIN, M. P. 2002. Spatial prediction of species distribution: an interface between ecological theory and statistical modelling. Ecological Modelling 157:101-118.

BEAUMONT, L. J., A. J. PITMAN, M. PAULSEN, AND L. HUGHES. 2007. Where will species go? Incorporating new advances in climate modeling into projections of species distributions. *In* G. C. Biology, editor. doi.10.1111/j.1365-2486.2007.01357.x.

BREIMAN, L., J. H. FRIEDMAN, R. A. OLSHEN, AND C. G. STONE. 1984. Classification and regression trees. Wadsworth International Group, Belmont, California, USA.

BUSBY, J. R. 1991. BIOCLIM - A bioclimate analysis and prediction system. Pages 64-68 *in* C. R. M. a. M. P. Austin, editor. Nature conservation: cost effective biological surveys and data analysis. CSIRO, Melbourne.

CARPENTER, G., A. N. GILLISON, AND J. WINTER. 1993. Domain - a flexible modeling procedure for mapping potential distributions of plants and animals. Biodiversity and Conservation 2:667-680.

DE'ATH, G. 2007. Boosted trees for ecological modeling and prediction. Ecology 88:243-251.

ELITH, J., C. H. GRAHAM, R. P. ANDERSON, M. DUDIK, S. FERRIER, A. GUISAN, R. J. HIJMANS, F. HUETTMANN, J. R. LEATHWICK, A. LEHMANN, J. LI, L. G. LOHMANN, B. A. LOISELLE, G. MANION, C. MORITZ, M. NAKAMURA, Y. NAKAZAWA, J. M. OVERTON, A. T. PETERSON, S. J. PHILLIPS, K. RICHARDSON, R. SCACHETTI-PEREIRA, R. E. SCHAPIRE, J. SOBERON, S. WILLIAMS, M. S. WISZ, AND N. E. ZIMMERMANN. 2006. Novel methods improve prediction of species' distributions from occurrence data. Ecography 29:129-151.

ESRI. 2004. ArcGIS 9.1. *in* ESRI, Redlands, CA.

FRIEDMAN, J. H. 2001. Greedy function approximation: a gradient boosting machine. Annals of Statistics 29:1189-1232.

GUISAN, A., A. LEHMANN, S. FERRIER, M. AUSTIN, J. M. C. OVERTON, R. ASPINALL, AND T. HASTIE. 2006. Making better biogeographical predictions of species' distributions. Journal of Applied Ecology 43:386-392.

HEIKKINEN, R. K., M. LUOTO, M. B. ARAUJO, R. VIRKKALA, W. THUILLER, AND M. T. SYKES. 2006. Methods and uncertainties in bioclimatic envelope modelling under climate change. Progress in Physical Geography 30:751-777.

HIJMANS, R. J., AND C. H. GRAHAM. 2006. The ability of climate envelope models to predict the effect of climate change on species distributions. Global Change Biology 12:2272-2281.

HIRZEL, A. H., J. HAUSSER, D. CHESSEL, AND N. PERRIN. 2002. Ecological-niche factor analysis: How to compute habitat-suitability maps without absence data? Ecology 83:2027-2036.

JARNEVICH, C. S., D. T. BARNETT, T. J. STOHLGREN, AND J. KARTESZ. 2007. A simple framework for an invasive species early warning system for counties. Frontiers in Ecology and the Environment in review.

MCCULLAGH, P., AND J. A. NELDER. 1989. Generalized linear models, 2nd edition. Chapman and Hall, London; New York.

PEARSON, R. G., AND T. P. DAWSON. 2003. Predicting the impacts of climate change on the distribution of species: are bioclimate envelope models useful? Global Ecology and Biogeography 12:361-371.

PETERSON, A. T. 2006. Uses and requirements of ecological niche models and related distributional models. Biodiversity Informatics 3:59-72.

PHILLIPS, S. J., R. P. ANDERSON, AND R. E. SCHAPIRE. 2006. Maximum entropy modeling of species geographic distributions. Ecological Modelling 190:231-259.

SOBERON, J., AND A. T. PETERSON. 2005. Interpretation of models of fundamental ecological niche and species' distributional areas. Biodiversity Informatics 2:1-10.

SSI. 2004. SYSTAT 11.0. San Jose, California, USA.

STOHLGREN, T. J., D. BARNETT, C. FLATHER, P. FULLER, B. PETERJOHN, J. KARTESZ, AND L. L. MASTER. 2006. Species richness and patterns of invasion in plants, birds, and fishes in the United States. Biological Invasions 8:427-447.

STOHLGREN, T. J., AND J. L. SCHNASE. 2006. Risk analysis for biological hazards: What we need to know about invasive species. Risk Analysis 26:163-173.

# PEST ERADICATION TECHNOLOGY – THE CRITICAL PARTNER TO PEST EXCLUSION TECHNOLOGY: THE MAUNGATAUTARI EXPERIENCE

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***Abstract:*** Invasive pest eradication is an increasingly viable management option for conservation and wildlife managers all over the world. The list of successful rodent eradications from isolated islands continues to grow globally. Now, with the development of effective pest exclusion technologies, the opportunities for eradicating multiple species of vertebrate pests from progressively larger fragments of habitat in mainland situations are also increasing. Attempts at eradicating up to fifteen separate species of pest mammal from indigenous forest fragments protected by Xcluder™ pest proof fencing have been made on the main islands of New Zealand. These include various assemblages of multiple species of deer (*Cervidae*), feral pig (*Sus scrofa*), brushtail possum (*Trichosurus vulpecula*), lagomorphs, rodents and a range of other predators. At some sites, individual species remain, while at others, complete eradication of a full suite of previously present pest species has been achieved. Using the 3,400 ha Maungatautari Project as a case study, we explore critical issues associated with successful pest eradication and reinvasion management in habitats protected by pest proof fencing, including eradication methods, finding and removing survivors, monitoring and surveillance, and the behaviour of reinvading pests. The transfer of the techniques successfully developed in New Zealand to other parts of the world is likely to depend as much on the regulatory environment at the management site as on the pest species present.

***Key Words:*** anticoagulant, brodifacoum, eradication, house mouse, invasive species, *Mus musculus*, pest-proof fencing, *Rattus rattus*, reinvasion, rodent.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Since the 1970s, introduced, invasive rodents have been eradicated from an increasing number of islands around New Zealand and, more recently, elsewhere in the world (Veitch and Bell 1990, Taylor et al. 2000, Towns and Broome 2003). Using these island techniques, there has been an increasing focus in New Zealand since 1999 to remove multiple species of pests from progressively larger fragments of mainland habitat. This approach has been encouraged primarily by new pest-proof fencing technologies that stop pests from reinvading (e.g., Day and MacGibbon 2007). While there are a number of independent projects underway in New Zealand, such as the Karori Wildlife Sanctuary, the Xcluder™ Pest Proof Fencing Company specifically has completed more than 20 fencing projects ranging from <1 ha to 3,400 ha. Of these, 11 projects have eradication of all pest species present inside the fenced area as one of the primary goals. A list of these projects, the techniques used to eradicate the pest species present and current project status in terms of pest eradication success within pest-proof fenced areas appears in Table 1. Figure 1 shows the location of the projects described.

Each of the projects detailed in Table 1 is run independently by different community groups or agencies using a variety of methods, service delivery contractors and/or pest eradication approaches. In this paper, we provide an overview of the Maungatautari Project (www.maungatrust.org), which has been the largest and most complex project to date and focus on important lessons learned to date from this ambitious project.

## BACKGROUND

Maungatautari is a highly eroded, andesitic volcanic cone in the central Waikato in the North Island of New Zealand. Although surrounded by farmland, the mountain itself (300-800 m asl) is mostly forest-covered, with about 3,400 ha (8,400 acres) of dense, primary, podocarp-broadleaved

**Table 1.** Summary of current pest-proof fencing projects in New Zealand where multi-species eradication has or will soon be attempted.

| Project Site | Size (ha) | Date Eradication Began | Methods* | No. of Pest** Species Present Originally | Pest Species Remaining at July 2007 |
|---|---|---|---|---|---|
| Karori (not Xcluder™ fence) | 250 | September 1999 | - Aerial bait<br>- Trapping<br>- Ground bait<br>- Bait stations | 10 | - Mice |
| Warrenheip | 16 | November 1999 | - Trapping<br>- Bait stations | 11 | - Mice |
| Rapanui Point | 1 | June 2003 | - Trapping<br>- Ground bait | 7 | - Nil |
| Riccarton Bush | 7 | July 2004 | - Trapping<br>- Bait stations | 7 | - Nil |
| Maungataurai (2 x Pilot Exclosures) | 100 (35 & 65) | September 2004 | - Aerial bait<br>- Trapping<br>- Ground bait<br>- Bait stations | 11 | - Nil |
| Tawharanui | 660 | September 2004 | - Aerial bait<br>- Trapping<br>- Ground bait<br>- Bait stations | 11 | - Mice |
| Young Nicks Head | 30 | July 2005 | - Trapping<br>- Ground bait<br>- Bait stations | 7 | - Mice (reinvasion) |
| Bushy Park | 98 | August 2005 | - Aerial bait<br>- Trapping<br>- Bait stations | 11 | - Nil |
| Macrae's Flat | 22 | August 2006 | - Trapping<br>- Ground bait | 7 | - Mice |
| Horseshoe Bay, Stewart Island | 160 | September 2006 | - Trapping<br>- Ground bait<br>- Bait stations | 7 | - Ship rats<br>- White-tailed deer |
| Maungatautari (Main Mountain) | 3,300 | November 2006 | - Aerial bait<br>- Trapping<br>- Ground bait<br>- Bait stations | 15 | - Mice<br>- Goats<br>- Hares<br>- Rabbits |
| Lake Rotakare | 227 | August 2007 | - Aerial bait<br>- Trapping<br>- Ground bait<br>- Bait stations | 12 | Eradication not yet undertaken |
| Orokonui | 248 | August 2007 | - Aerial bait<br>- Trapping<br>- Ground bait<br>- Bait stations | 12 | Eradication not yet undertaken |

* Aerial baiting includes helicopter application of brodifacoum cereal pellets. A wide range of traps and toxins are also registered for use in New Zealand.

** Refer to Table 2 for a list of typical pest species found in New Zealand project areas.

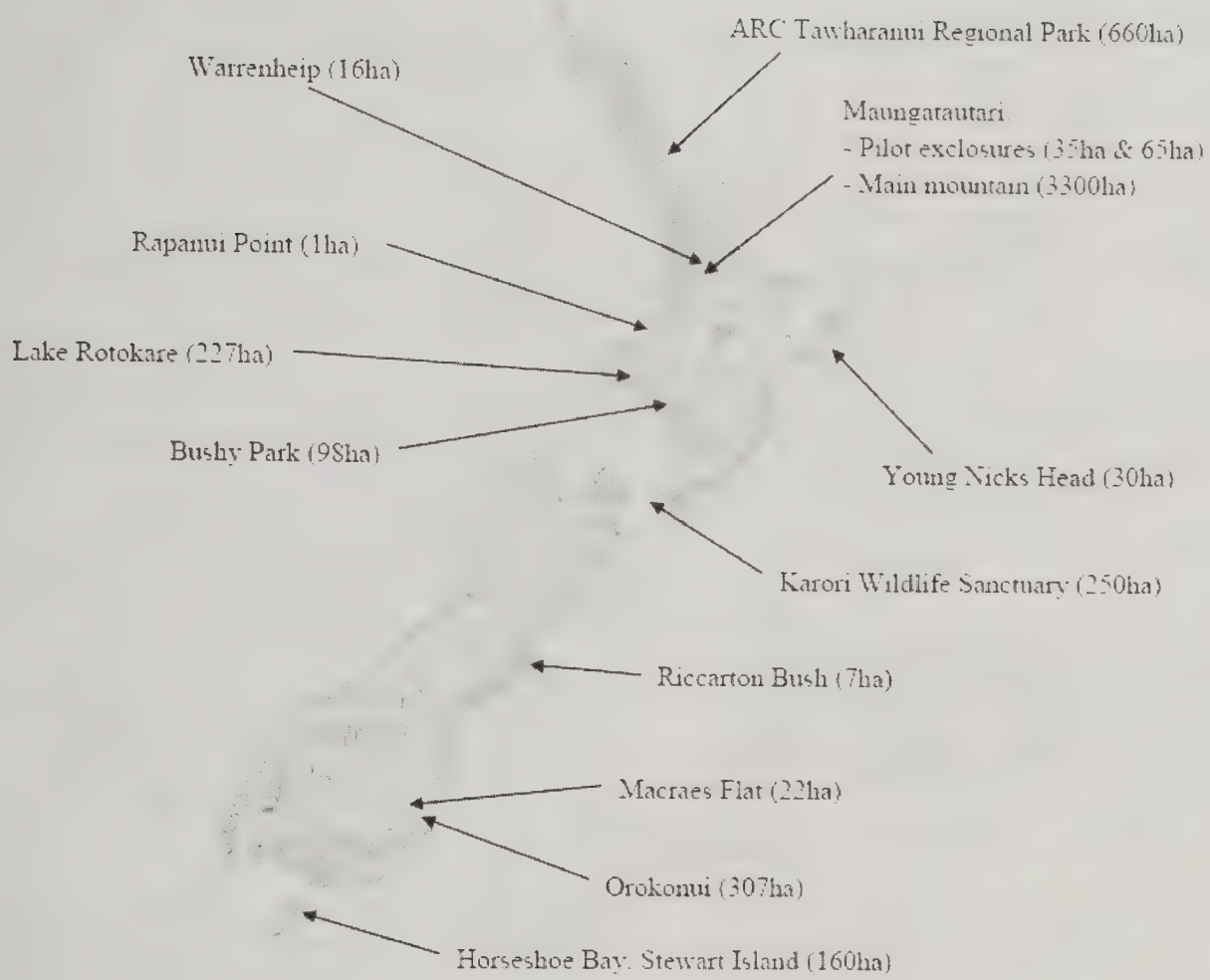



**Figure 1.** Current pest-proof fencing projects in New Zealand where multi-species eradication has or will soon be attempted ([Xcluder™ 2006], also refer to Table 1).

native forest. The Maungatautari Ecological Island Trust has a vision to: "...remove, forever, introduced mammalian pests and predators from Maungatautari, and restore to the forest a healthy diversity of indigenous plants and animals not seen in our lifetime..."

The Trust's approach has been to completely ring fence the mountain with 47 km of Xcluder™ pest-proof fence and then attempt to eradicate 15 species of pest mammal species (Table 2) within the 3,400 ha fenced area. Once this is achieved, ecological recovery of the mountain is anticipated and a wide range of pest-vulnerable indigenous species, now extinct on the mountain, can begin to be reintroduced.

As a pilot for the larger mountain, two smaller exclosures were constructed protecting 35 ha (Northern Exclosure) and 65 ha (Southern Exclosure) respectively (Figure 2). Eradication began in the two pilot exclosures in September 2004. Information gained from the eradication attempts within the pilot exclosures has formed the basis for a larger scale eradication attempt on the main mountain, which began in 2006.

## ERADICATION METHODS

The primary tool used to eradicate most small mammal pest species on Maungatautari has been the aerial application of cereal pellets containing the anticoagulant toxin brodifacoum at a

**Table 2.** Species list of typical vertebrate pests present in pest-proof fencing project areas of New Zealand.

| Common Name | Latin Name |
|---|---|
| Cat | *Felis catus* |
| Fallow deer | *Dama dama* |
| Ferret | *Mustela furo* |
| Goat | *Capra hircus* |
| Hare | *Lepus europaeus occidentalis* |
| Hedgehog | *Erinaceus europaeus occidentalis* |
| Mouse | *Mus musculus* |
| Norway rat* | *Rattus norvegicus* |
| Pig | *Sus scrofa* |
| Possum | *Trichosurus vulpecula* |
| Rabbit | *Oryctolagus cuniculus cuniculus* |
| Red deer | *Cervus elephus* |
| Ship rat | *Rattus rattus* |
| Stoat | *Mustela erminea* |
| Weasel | *Mustela nivalis vulgaris* |

* not confirmed but present near-by

concentration of 20 ppm, marketed by its New Zealand manufacturers, Animal Control Products Ltd, as "PestOff Rodent Bait 20R". However, the registration for this product using aerial application methods is very restrictive due to the residual nature of the toxin in the food chain. A special Code of Practice (CoP), which complied with the New Zealand Government's Agricultural Compounds and Veterinary Medicines Act (1996), had to be developed to ensure the risks of livestock contamination and other human health risks were satisfactorily managed, before the New Zealand Food Safety Authority would approve aerial application of PestOff Rodent Bait 20R for use in mainland situations. A copy of the CoP is available from the Animal Control Products website at www.pestoff.co.nz. The Maungatautari aerial bait application operations also required a Resource Consent under the Resource Management Act (1991) which set out strict conditions for the discharge by air, of a contaminant to land or water. These conditions included stakeholder consultation, extensive water monitoring and non-target impact reporting (Environment Waikato 2004, 2006).

In previously successful offshore island rodent eradications, two applications of brodifacoum bait have been routinely used, with bait sowing swathes being overlapped by 50% to ensure complete bait coverage. Bait application is typically timed for the late winter/early spring period when bait acceptance is highest (Gillies et al. 2003), and rodent numbers are lowest after winter mortality and before spring-summer breeding (Innes et al. 1995, Innes 2005, Ruscoe and Murphy 2005). On Maungatautari, there were a number of variations to the standard island rodent eradication bait application technique:

1. In both pilot and main mountain eradication attempts, there was a significant increase in bait sowing rates from the typical 8 kg/ha and 4 kg/ha for the initial and follow-up operations to 15 kg/ha and 10 kg/ha respectively, due to the presence of a large number of pest species that would find the baits highly palatable. While the list of pest species included ship rats (*Rattus rattus*) and house mice (*Mus musculus)* that would eat few baits, the slow acting nature of the toxin combined with the presence of the larger brushtail possums, hedgehogs (*Erinaceus europaeus occidentalis)*, hares (*Lepus europaeus occidenalis)*, rabbits (*Oryctolagus cuniculus cuniculus)* and pigs meant bait could quickly be consumed from more favourable habitat sites, resulting in some pests being unable to access bait, or receiving sub-lethal exposure. Ensuring adequate bait penetration through a dense, complex forest vegetation cover on the mountain was also a consideration in increasing bait sowing rates.

2. For standard island rodent eradications, the second aerial bait application is typically less than 10 days after the first application based on a time to death of 3 to 7 days for rodents. The average time to death after consuming a lethal dose of brodifacoum for the range of animals present on Maungatautari ranges from 5 days (rodents) to 22 days (possums) (Eason and Wickstrom 2001), so the second aerial bait application for both pilot and main mountain eradications occurred six weeks after the first, to allow larger animals that had consumed bait a chance to die before the second aerial bait application occurred.

3. For the main mountain, a third aerial bait application targeting surviving mice proved necessary the following winter (June-July 2007, 6 months after the second operation), again at a sowing rate of 10 kg/ha.

4. All aerial bait applications were accompanied by hand-broadcast bait application immediately inside the fence to reduce the chances of bait being

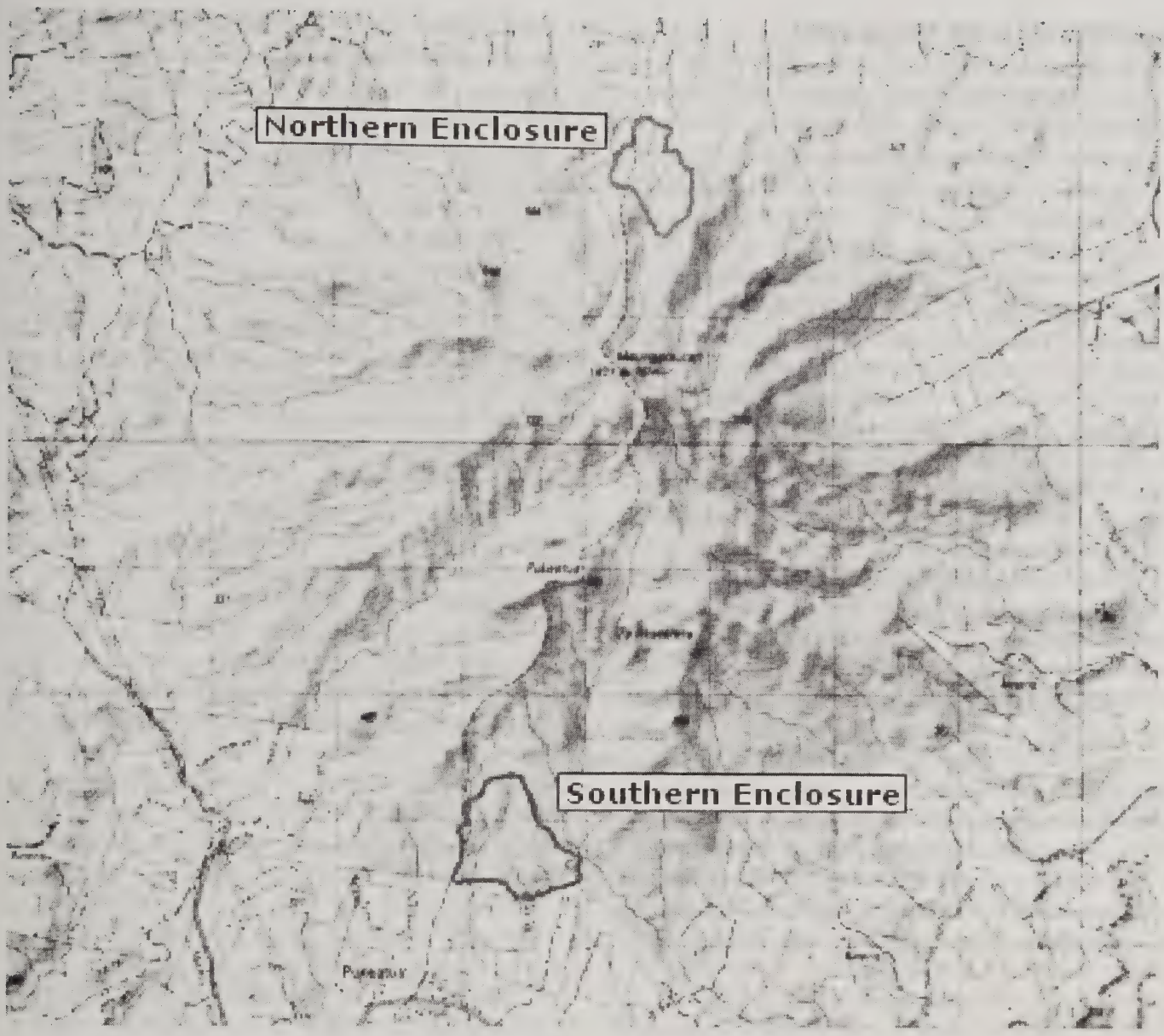



**Figure 2.** Map of Maungatautari Project area.

over-flown from the helicopter onto adjoining grazing land, and to ensure complete bait coverage.

Secondary poisoning has been well documented for predators feeding on the carcasses of other pests that have been poisoned as primary targets in New Zealand pest control operations using brodifacoum and mono-sodium fluoroacetate (1080) (Alterio 1996, Alterio et al. 1997, Alterio and Moller 2000, Department of Conservation 2006). While it was anticipated that the impact of the aerial bait applications on prey species would seriously impact predators through secondary poisoning, follow-up trapping and or poisoning was anticipated for most pest species, to be guided by the results of post aerial bait application monitoring.

Pigs, goats (*Capra hircus*), and deer (*Cervus elephus and Dama dama*) evacuated the pilot treatment areas during fence construction so no information about these pests was available for the main mountain from the pilot eradication attempt. However, effective goat and deer eradication techniques are already well developed in New Zealand (Parkes 1990), and these species were not considered a major issue. Pigs were considered to be extremely vulnerable to brodifacoum poisoning.

## TARGET PEST MONITORING

### Pilot Exclosures

The use of ink-tracking tunnel networks (King and Edgar 1977, Ratz 1997) for monitoring small pest mammal populations has been critical for both the pilot and main mountain eradication attempts on Maungatautari.

The relatively small, accessible nature of the pilot exclosures allowed tracking tunnel grids of 25 m x 25 m and 50 m x 50 m to be established in the 35 ha and 65 ha Northern and Southern exclosures respectively, to detect surviving pests. The 25 m x 25 m tracking tunnel grid was originally thought to be particularly important to detect surviving mice. Some sampling using these networks occurred prior to any bait application to help train personnel and to provide some assessment of pre-operational pest presence/densities.

After the second aerial bait application, all tracking tunnels were monitored weekly for the first year, then monthly until six months of zero mammal tracking was achieved. Tunnels were serviced by teams of volunteers who replaced cards to allow expert interpretation of any tracking, and re-baited tunnels with fresh peanut butter. Once eradication was confirmed, clean cards and fresh peanut butter has been placed for one 7 to 10 day period every three months before cards are uplifted for interpretation. Fence breach monitoring and surveillance protocols have also been established for situations where tree falls or other fence damage, or a range of other scenarios, result in the opportunity for pest reinvasion.

When a ship rat, house mouse or hedgehog was detected on the tracking tunnel grids, it was targeted with snap traps set in the tracking tunnels using peanut butter on the trigger plate for rodents, or Mark IV Fenn traps baited with canned cat food for hedgehogs.

Starting six months after the second aerial bait application, a biomarker (rhodamine B) was distributed on non-toxic cereal pellets around the outside of the pilot exclosures to bio-mark the resident rodents. This was undertaken in an attempt to help definitively distinguish any animals caught on the inside of the pilot exclosures as either reinvaders or survivors. These pellets were distributed at 6-8 week intervals for 18 months.

Following increased house mouse tracking in the Southern Exclosure in March 2006, bait stations containing poison blocks with 20 ppm brodifacoum were deployed at all sites that tracked mice, until tracking stopped. Some of these bait stations were lured with mouse urine.

### Reinvasion Risk Assessment

To begin understanding the risk of pest animals reinvading after damage to a pest-proof fence (e.g., a tree fall), research to quantify the detection and use of fence breaches by pests resident outside the fence was initiated. Using a 10 km section of the larger Maungatautari pest proof fence where eradication had not yet begun, 25 artificial holes (to simulate fence mesh damage) and 25 artificial ramps (to simulate a fallen tree branch leaning on the fence) were created along the fence. Unbaited tracking cards were placed at each artificial breach in such a way that all use of the breaches by pest animals from outside the fence could be detected and the species identified. The artificial breaches were then inspected for pest activity daily for the first 4 days and once after 4 weeks.

### Reinvasion Behavior Research

To further understand the behaviour of potentially reinvading rodents, and to help develop effective reinvasion response plans, a small research programme was begun following removal of what was thought to be the last surviving rat from inside the Southern Exclosure. Six ship rats, caught against the outside of the fence, were radio collared and released into the rat-free exclosure during April-September 2006. Ship rats were selected because they are common outside the fence (probably 3-7 per ha), arboreal, active all year round and highly exploratory, and, therefore, one of the most likely species to reinvade. Only adult, territorial males were used and the animals were not de-sexed to allow for as natural behaviour as possible. The rats were released next to their existing home ranges, thus mimicking the most likely natural invasion via a fence breach or a falling tree. Rats were introduced sequentially so that only one translocated rat was ever present in the exclosure at any time. Key objectives were to describe the movement and denning behaviour of such invasives, and to see if the standard invader detection system (the 50 m x 50 m tracking tunnel grid) detected the rats. Our intention was to track each rat until its movements stabilised, when it would be removed, although in practise rat movements within the exclosure did not stabilise. We tracked the rats nightly, from evening for about 5 hours, for a week after release, then more sporadically but at least every few days.

**Main Mountain**

On the main mountain, perimeter access track networks and cut ridgeline/valley transects allowed for the establishment of over 1,700 tracking tunnels spaced at 100 m intervals across the mountain. If each tunnel can attract and record a mammal within a 100 m radius, then this network provides coverage of 57% of the project area. Some transects within a representative range of habitats across this network were assessed prior to the start of the eradication attempt to help understand pre-operational rodent densities.

Since the second aerial bait application, this network has been monitored on a two-monthly cycle where clean cards and fresh bait (both peanut butter and minced rabbit meat) are placed for one 7-10 day period before cards are uplifted for reading.

Likely pig survival was indexed using six young (18 to 25 kg) wild pigs, carrying radio-transmitter ear tags with a mortality function. The pigs were released onto the mountain 4 months prior to any aerial bait application and were located weekly until the first bait application, then daily after that.

## RESULTS

**Pilot Exclosures**

House mice were virtually undetectable in the exclosures prior to aerial bait applications, with just 4% tracking in the Northern Exclosure and no detections in the Southern Exclosure. Ship rat density was extremely high with pre-operational tracking rates of 98% and 96% in the Northern and Southern exclosures, respectively. Possums, hares and rabbits were present in low numbers but no carnivores (mustelids or cats, *Felis catus*) were detected. It is possible that these animals emigrated over the fence in the months between fence closure and the beginning of the eradication (6 months and 2 months in the Northern and Southern exclosures, respectively), and that their absence boosted ship rat numbers. Both mustelids and cats were tracked immediately outside the exclosure fences at the time of these eradication attempts.

A total of 7 ship rats, at least 5 house mice and 1 hedgehog were detected and removed by trapping in the 20 months following the second bait application in the two pilot exclosures, although further mice were almost certainly poisoned by brodifacoum baits in bait stations deployed in the Southern Exclosure starting March 2006 to specifically target increasing mouse detections. No other species are known to have survived the two aerial bait applications, although a single hare was seen once within two weeks of the first bait application, but subsequently disappeared.

All animals trapped after aerial bait application were necropsied to provide information on sex, age, reproductive status and toxin residues. Table 3 details all animals removed by follow-up trapping in the pilot exclosures post aerial bait application.

Some surviving ship rats were detected almost immediately and were relatively easy to catch at tracking tunnels once they had tracked. However, some rats remained undetected for up to 18 months before being tracked and then trapped. Rat home ranges appeared to increase significantly over time with earliest rat detections covering 2 to 3 ha (based on tracking tunnel detections) while in the last detection, a female covered almost 20 ha.

Surviving mice did not show up in the pilot exclosures until 6 months after the second aerial bait application (and then only in the Southern Exclosure). Home ranges of surviving house mice post eradication ranged from 2-6 ha, much larger than ranges in higher density mouse populations (Ruscoe and Murphy 2005).

The single surviving hedgehog tracked tunnels over 10.5 ha before being caught.

Several lines of evidence lead to the conclusion that these animals were all survivors and not reinvaders, despite the sometimes long period (up to 18 months) before detection. No Rhodamine B dye was present in any of the animals trapped within the Southern Exclosure; all animals trapped in both exclosures had brodifacoum residues and no females were lactating or pregnant, despite being of mature age and in good condition.

Two possible reinvasion events within the Southern Exclosure were detected where rat prints were found within a few days of a tree fall on the fence. In both cases, despite reinvasion responses by project staff, the rats were subsequently neither trapped nor tracked. It is possible that they emigrated back outside the fence, as did most of the experimentally translocated ship rats (see below).

Based on movements of the animals we detected, the intensive 25 m x 25 m tracking tunnel grids did not prove necessary to detect even mice surviving the aerial bait applications, with 50 m x 50 m spacings proving more than sufficient to detect all rat, mice and hedgehog survivors. In fact, a tracking tunnel layout on a 100 m x 100 m grid system is likely to have been sufficient to intercept the home ranges of all animals caught subsequent to the aerial bait applications in the pilot

**Table 3.** Summary of mammals trapped from Maungatautari pilot exclosures - post aerial bait application (Sept/Oct 2004).

| Site/Species | Timing | Sex | Range | Reproductive Status | Rhodamine' B'? | Toxin Residue? |
|---|---|---|---|---|---|---|
| 35 ha (Northern) Pilot Exclosure: | | | | | | |
| Ship rat | Nov 2004 | Female | 2.5 ha | Never bred | N/A* | Positive |
| Hedgehog | Dec 2004 | Female | 10.5 ha | Never bred | N/A* | Positive |
| Ship rat | Jan 2005 | Male | 5.0 ha | Testes abdominal | N/A* | Positive |
| Ship rat | Apr 2005 | Female | 3.0 ha | Never bred | Negative | Positive |
| 65 ha (Southern) Pilot Exclosure: | | | | | | |
| Ship rat | Nov 2004 | Female | 3.0 ha | Never bred | N/A* | Positive |
| Ship rat | Nov 2004 | Female | 2.0 ha | Never bred | N/A* | Positive |
| Ship rat | Jan 2005 | Female | | Never bred | N/A* | Positive |
| Mouse | Apr 2005 | Male | 3.0 ha | Testes descended | N/A* | Positive |
| Mouse | Sept 2005 | Male | 6.0 ha | Testes descended | Negative | Positive |
| Ship rat | Mar 2006 | Female | 19.5 ha | Never bred | Negative | Positive |
| Mouse | Apr 2006 | Female | 1.0 ha | Pregnant – 7 embryos | Negative | Positive** |
| Mouse | May 2006 | Male | 1.5 ha | Testes descended | Negative | Positive** |
| Mouse | May 2006 | Female | 1.0 ha | Juvenile | Negative | Not tested** |

* Rhodamine B baiting outside pilot exclosures began May 2005.
** Brodifacoum present in bait stations from April 2006.

exclosures and a 100 m x 200 m layout would have detected the majority.

Both pilot exclosures are now considered completely pest free, the Northern Exclosure since April 2005 and the Southern Exclosure since July 2006.

**Reinvasion Risk Assessment**

Of 50 artificial breaches placed along a 10km section of pest proof fence, 4 (8%) were used within the first 24 hours (brushtail possums over 2 ramps; mice through 2 holes). The exact time of use was not determined. Over the following 3 days, 11 (22%) of the 50 breaches were used, with 8 of these breaches used on at least 2 nights. Cats, brushtail possums and rats (species not determined) crossed the fence via ramps and mice and rats used the artificial holes. After 4 weeks, 32 of 45 breaches (71%; some tracking cards were missing from breaches so were not included in the data) were used. Possums, cats, rats and mice used the artificial ramps. Mice, rats, hedgehogs and rabbits were tracked using the artificial holes in the fence. Although mustelids (ferrets *Mustela furo* and stoats, *Mustela erminea*) were observed and trapped along the fence line during the period of this experiment, none were detected on the tracking cards to be using the artificial ramps or holes.

**Reinvasion Behaviour Research**

Table 4 details the time spent inside the exclosure, the maximum distance travelled, the number of tracking tunnels used and the fate of the six male ship rats released in April-October 2006. Unexpectedly, four of the six rats sooner or later climbed over the fence and returned to their original home ranges. The remaining two were eventually killed by brodifacoum poison laid for mice. The rats that did move considerably inside the exclosure all showed a similar pattern to each other. They stayed within 100 m of the release point for about 3 days, usually denning at the same place, but we cannot tell if this was related to recovery from anesthesia and radio-collaring, or

**Table 4.** Summary of the behaviour of six radio-collared, adult, male ship rats released within the 65 ha (Southern) Maungatautari exclosure.

| Rat No. | Time In Exclosure | Tracking Tunnels Tracked | Largest Distance Travelled From Release Point | Fate Of Animal |
|---|---|---|---|---|
| 1 | 7 days | 2 | 650 m | Died within exclosure from brodifacoum laid for mice |
| 2 | 31 days | 7 | 1000 m | Died within exclosure from brodifacoum laid for mice |
| 3 | 3 days | 0 | 20 m | Trapped outside exclosure in original home range |
| 4 | 7 days | 8 | 600 m | Returned to original home range outside exclosure; final fate never verified. |
| 5 | < 24 hrs | 0 | 20 m | Returned to original home range outside exclosure. Verified dead after first main mountain aerial poison drop |
| 6 | < 6 hrs | 0 | 20 m | Returned to original home range outside exclosure; final fate never verified. |

reflected true behaviour of a cautious rat invading new territory. They then gradually made bigger movements into the exclosure, eventually changing den sites as well. Movements were undeniably much larger that those usually made by ship rats in normally structured NZ populations. They used similar routes on consecutive nights, and the same dens, perhaps implying that they were using scent trails. Only three of the six rats tracked were detected at any tracking tunnels at all. The rat that stayed longest (31 days) in the exclosure tracked at only seven tunnels.

**Main Mountain**

Rats were detected at 48% of the sample tracking tunnel transects across a representative range of habitats prior to the start of the eradication attempt, while house mice were detected at 18% of the tracking tunnels sampled, mostly in habitats with dense ground cover (grass, gorse, windrows) where rat tracking was very low or zero. Very limited tracking of other pest mammals was detected with isolated possum, cat and stoat tracks observed. This does not mean that these species were scarce, however, since the tracking tunnels used were baited with peanut butter to primarily target rodents. Hedgehogs hibernate in winter which is when these assessments were made.

Rats were first seen dead after day three of the first aerial bait application on the main mountain. The first radio-tagged pig died on day seven and all six tagged pigs were dead by day eleven. All pigs had significant brodifacoum residues in their livers. The first dead cat was found at day eleven but this animal had been dead for several days. A live possum was observed about 5 weeks after the first aerial bait application suffering severe toxicosis and was euthanised. There were also reports of live cats, a single live hedgehog, numerous rabbits and at least two hares up to five weeks after the first aerial bait application.

The need for suitable weather conditions meant that the second aerial bait application occurred six weeks after the first. Isolated sightings of rabbits, hares and cats continued inside the perimeter fence around the mountain, even after the second bait application.

The first tracking tunnel assessment across the main mountain was undertaken one month after the second bait application. This assessment detected house mice on 0.92% of tunnels, at three locations. Cats were detected on 3.8% of tracking tunnels, at two broad locations, though both appeared to be

covering large ranges along several kilometers of the internal perimeter of the pest-proof fence. Two male cats were subsequently trapped at these locations and tracking stopped.

Additional rabbit poisoning, using hand laid 20 ppm Pindone pellets (a low-toxicity, first generation anticoagulant specially formulated in cereal pellet form and marketed for rabbit control in New Zealand by Pest Management Services Ltd.), was undertaken at four sites where rabbits continued to be seen. One hare was shot, but hares continued to be seen at two locations. Mouse trapping began at all three detection locations and mice were caught at all sites. However, in the subsequent tracking tunnel assessment in March 2007, while no other mammal detections were recorded, mice were detected at 7.28% of tunnels and large-scale hand spreading of PestOff Rodent Bait 20R was undertaken at a range of these sites. By the May 2007 tracking assessment, 4.93% of tunnels across the mountain were still detecting mice, although the lack of other mammal detections continued.

A third aerial bait application targeting mice is planned for winter 2007 across the whole mountain at a sowing rate of 10 kg/ha. The results of that aerial third aerial bait application remain unknown at the time of writing this paper.

## DISCUSSION

The Maungatautari eradication case study, together with evidence from a range of other pest-proof fence project sites in New Zealand, shows that multi-species pest mammal eradication can be achieved behind pest-proof fences in mainland situations, at least on small to medium landscape scales. The aerial application of brodifacoum-based cereal pellet baits, if well planned and conducted, provides an important 'first shot' that can lay the foundation for a successful eradication. Issues such as the suite of pest species present, their density and interspecific relationships, the nature of the habitat, timing, bait sowing rates and bait coverage are all important planning issues to be carefully considered for aerial bait applications.

Even with a well planned aerial bait application, small numbers of survivors, particularly house mice, should be anticipated. It is not clear from the existing attempts exactly why mice have been successfully eradicated in some projects and not in others. However, the common survival of mice is seemingly a result of dense habitat types; e.g., long grass which may restrict access to bait, but it is also likely to be behavioural, related to the presence of ship rats. Ship rats appear to act as both predators and competitors of mice and, as such, often mask the presence of mice, especially when rats occur at high density within project areas. Mice may not appear in any significant numbers, or at all, until ship rats are reduced to extremely low (near zero) levels or even removed completely. Careful planning is required where these two species are known to co-exist, but additional research is still badly needed to refine the menu of most effective tools and approaches for removing small, residual pockets of house mice.

Perseverance is critical in removing the last few individuals. Increasingly intensive monitoring is required as pest density approaches zero, although home range expansion of survivors at very low density can be expected. A range of eradication tools is required to target animals that survive the use of primary tools such as aerial bait application and follow-up trapping. Management agencies should have an expectation that a long-term commitment must be sustained and funded to ensure eradication is achieved, and to avoid projects being abandoned too early. In even small project areas, total eradication of a full suite of mammalian pest species may require intensive effort over a period of years before it is successfully achieved.

Reinvasion remains an ever present risk for pest-proof fence project areas. Monitoring and surveillance regimes are a critical part of ongoing management of such areas. Reinvasion response planning, preparedness and response capability are especially important due to the potentially rapid nature of reinvasion as a result of a fence breach.

Our initial pest reinvasion risk research at Maungatautari suggests that fence breaches need to be identified and rectified inside the first 24 hours if pest invasion events from multiple species are to be minimized. For projects where daily physical inspection is unachievable (such as the 47 km fence around Maungatautari), remote fence surveillance technology becomes an important additional management tool (Day and MacGibbon 2007). Ongoing research is now underway to better determine reinvasion risk for different pest species and identify suitable invasion risk management options for Maungatautari and other projects.

Our initial research with ship rats suggests that reinvaders of this species may spend at least the first few days at the entry point, again reinforcing the need for rapid response as such animals are easier to target in a more confined area at this early

stage. The use of established scent trails by reinvading animals suggests that dogs could play an important role in reinvasion response. We are pursuing this line of research at Maungatautari.

The regulatory environment in New Zealand currently allows for the use of aerial application of brodifacoum bait subject to adherence to a strict Code of Practice and to Resource Consent conditions. The unique situation within New Zealand where there are no vulnerable, ground dwelling native mammals, is a key part of this regulatory environment. The presence of native mammals or other wildlife vulnerable to brodifacoum is likely to restrict such tools in many parts of the world, unless careful cost-benefit analysis can show overwhelmingly positive outcomes for the indigenous ecology from the use of such tools, to eradicate serious pest mammals.

## CONCLUSION

- Aerial application of brodifacoum provides a broad-spectrum pest mammal eradication tool, if used in conjunction with follow-up monitoring, trapping, shooting and poisoning to remove the limited numbers of survivors that may remain.
- A range of site specific variables must be considered when planning such aerial bait applications to maximize their effectiveness.
- Of the introduced mammal species we have worked with in NZ, house mice are the hardest to eradicate and can remain almost undetectable until ship rats have been removed or reduced to near zero density.
- Perseverance is critical in removing the last few individuals. Management agencies should have an expectation that a long-term commitment must be sustained and funded to ensure eradication is achieved.
- Rapidly identifying and rectifying breaches, coupled with reinvasion surveillance and monitoring are critical aspects of post eradication management. While baited tracking tunnels have been critically helpful at Maungatautari, little is known about the actual probabilities that they will detect all survivors or invaders of various species.
- We are now experimentally researching the comparative effectiveness of dogs and tracking tunnels at locating single survivors or invaders.
- The transfer of the techniques successfully developed in New Zealand, to other parts of the world, are likely to depend as much upon the regulatory environment at the management site as on the pest species present.

## LITERATURE CITED


ALTERIO, N. 1996. Secondary poisoning of stoats (*Mustela erminea*), feral ferrets (*Mustela furo*), and feral house cats (*Felis catus*) by the anticoagulant poison, brodifacoum. New Zealand Journal of Zoology 23:331-338.

ALTERIO, N., K. BROWN AND H. MOLLER. 1997. Secondary poisoning of mustelids in a New Zealand *Nothofagus* forest. Communications from the mammal society no. 75, Journal of Zoology 243:863-869.

ALTERIO, N., AND H. MOLLER. 2000. Secondary poisoning of stoats in a South Island podocarp forest, New Zealand: implications for conservation. Wildlife Research 27:501-508.

DAY, T. D. AND R. J. MACGIBBON. 2007. Multiple-species exclusion fencing and technology for mainland sites. Pages 418-433 *in* G. W. Witmer, W. C. Pitt, and K. A. Fagerstone, editors. Proceedings of the managing vertebrate invasive species. August 2007. Fort Collins, Colorado, USA.

DEPARTMENT OF CONSERVATION. 2006. Tongariro Kiwi Sanctuary 1080 pest control operation, September 2006, Post–operational report. Unpublished internal report, Department of Conservation, Turangi, New Zealand.

EASON, C. T., AND M. WICKSTROM. 2001. Vertebrate pesticide toxicology manual (poisons). Information on poisons used in New Zealand as vertebrate pesticides. Department of Conservation Technical Series 23. Department of Conservation, Wellington, New Zealand.

ENVIRONMENT WAIKATO. 2004. Resource Consent 111219. Discharge permit for discharge to land - aerially apply up to 3,000kg of the animal pesticide PestOff, containing brodifacoum, for mammalian pest eradication purposes within two pest proof exclosures at Maunagatautari.

ENVIRONMENT WAIKATO. 2006. Resource Consent 113994. Discharge permit for discharge to land - aerially apply the animal pesticide Pestoff containing brodifacoum in three applications at a maximum rate of 20 kg/ha for mammalian pest eradication purposes.

GILLIES, C. A., J. CAMPBELL, N. MARSH AND M. GEMBITSKY. 2003. Seasonal differences in bait acceptance by forest dwelling rats following simulated aerial 1080 possum control operations in New Zealand: interim results. Pages 343-345 *in* Singleton et al., editors. Rats, mice and people: rodent biology and management. Australian Centre for International Agricultural Research, Canberra, Australia.

INNES, J. 2005. Ship rat. Pages 187-203 *in* C. M. King, editor. The handbook of New Zealand mammals. Oxford University Press, Melbourne, Australia.

INNES, J., B. WARBURTON, D. WILLIAMS, H. SPEED, P. BRADFIELD. 1995. Large-scale poisoning of ship rats (*Rattus rattus*) in indigenous forests of the North Island, New Zealand. New Zealand Journal of Ecology 19:5-17.

KING, C. M. AND R. J. EDGAR. 1977. Techniques for trapping and tracking stoats (*Mustela ermine*); a review and a new system. New Zealand Journal of Zoology 4:193–212. Maungatautari Ecological Island Trust, www.maungatrust.org.

PARKES, J. P. 1990. Eradication of Feral Goats on Islands and Habitat Islands. Journal of the Royal Society of New Zealand 20:297–304.

RATZ, H. 1997. Identification of footprints of some small mammals. Mammalia 61:431-441.

RUSCOE, W. A., AND E. C. MURPHY. 2005. House mouse. Pages 204-221 *in* C. M. King, editor. The handbook of New Zealand mammals. Oxford University Press, Melbourne, Australia.

TAYLOR, R. H., G. W. KAISER, AND M. C. DREVER. 2000. Eradication of Norway rats for recovery of seabird habitat on Langara Island, British Columbia. Restoration Ecology 8:151-160.

TOWNS, D. R., AND K. G. BROOME. 2003. From small maria to massive campbell: forty years of rat eradication from New Zealand islands. New Zealand Journal of Zoology 30:377-398.

VEITCH, C. R., AND B. D. BELL. 1990. Eradication of introduced animals from the islands of New Zealand. *In* D. R. Towns, C. H. Daugherty, I. A. E. Atkinson, editors. Ecological restoration of New Zealand islands. Conservation Sciences Publication 2.

Xcluder™ . 2006. Xclude It!, Issue 1 – June 2006. Xcluder™ Pest Proof Fencing Company, Cambridge, New Zealand.

# MANAGEMENT OF INVASIVE VERTEBRATES IN THE UNITED STATES: AN OVERVIEW

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***Abstract:*** At least 161 introduced/invasive vertebrates have become established in the United States and its territories, including at least 81 mammalian, 94 avian, and 86 reptilian/amphibian species. Particularly problematic species include feral cats and dogs; feral pigs; commensal rats and mice; starlings, pigeons, and house sparrows; and bullfrogs, brown treesnakes, and coqui frogs. We briefly review these introductions and the types of damage they cause. We review the basic types of methods used for control or eradication of each taxonomic group, including physical, chemical, biological, and cultural methods. We also discuss some of the challenges in managing these species, including issues with the use of toxicants, land access, public attitudes, and monitoring difficulties. Finally, we discuss some ongoing research and future research needs, including fertility control, improved detection methods, improved attractants, improved barriers, improved capture methods, and risk assessment methods.

***Key Words:*** animal damage, eradication, introduced species, invasive species, management.

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## INTRODUCTION

Vertebrate species have been introduced to almost all parts of the world for thousands of years. The large volume of worldwide trade and transportation have accelerated the rate of introductions in the last 150 years or so. Animals are introduced for many reasons, both purposeful and accidental. Purposeful introductions occur for food and fur, work animals, sport hunting and fishing, companion animals, aesthetics, and pest control. Accidental introductions occur because of stowaways in transport vehicles, released animals (without an original intent to release), escapees, and, in some cases, because of range expansion of a species (often facilitated by human activities and land uses).

While many introduced vertebrate species have provided important resources and economic gains for humans and many do not cause undue adverse effects (especially with appropriate management), some have increased their distribution and have caused serious adverse effects. These include disease and safety hazards, predation and competition with native species, crop consumption and contamination both in the field and during storage, livestock predation, and, in some cases, significant environmental degradation. Unfortunately, for many species of introduced vertebrates, we do not yet know if they are causing, or in the future will cause, significant harm to the environment or human resources. Hence, our use of the term "introduced/invasive" vertebrate species.

Lists of mammalian (Long 2003) and avian (Long 1981, Lever 1987) introductions around the world have been compiled with relatively minor discussion of impacts and control efforts. As political and social awareness has grown, efforts to assess introduced species and their adverse impacts have increased (Pimentel et al. 2005). A conference held in New Zealand in 2001 brought attention to the widespread problems with invasive species and illustrated many of the diverse projects being conducted around the world to try to resolve some of the situations (Veitch and Clout 2002). Witmer and Lewis (2001) reviewed vertebrate introductions in the Pacific Northwestern Region of the United States (US), the resultant impacts, state and federal legislation, management methods and strategies, and some case studies for birds, mammals, and amphibians/reptiles (herptiles). At the 12th Wildlife Damage Management Conference in Traverse City, Michigan (2005), speakers reviewed issues of introduced birds (Avery and

Tillman 2005), herptiles (Pitt et al. 2005), ungulates (Lowney et al. 2005), carnivores (Witmer et al. 2005), and rodents (Witmer and Eisemann 2005). Pitt and Witmer (2007) reviewed issues associated with invasive predators across several taxa.

In this paper, we compile and update lists of introduced vertebrates (mammals, birds and herptiles) in the US (mainland and Hawaii). We present some of the most problematic species and the types of adverse impacts they cause. We also list the main management methods and tools used to control and eradicate invasive vertebrates in the US. Finally, we discuss some of the remaining challenges in addressing invasive vertebrate management in the US and some research needs. Scientific names are presented in the tables and are not repeated in the text.

## VERTEBRATES INTRODUCED IN THE UNITED STATES

We compiled our lists of introduced vertebrate species in the US from a variety of sources, including several publications (e.g., Bury and Luckenbach 1976, Long 1981, Mooney and Drake 1986, Moulton and Pimm 1986, Lever 1987, Hawaii Audubon Society 1989, American Ornithologist's Union 1998, Witmer and Lewis 2001, Long 2003, Teer 2003, Meshaka 2006, Kraus 2007), but also from several state wildlife agency and conservation organization websites (available upon request). Our focus was on the US mainland and Hawaii, so we have not included some of the species introduced to US territories such as the US Virgin Islands, Puerto Rico, and Guam. An important exception is the inclusion of the brown treesnake in Guam as it is a major problem invasive species receiving a considerable investment in research and control efforts. The list we have compiled is quite long because we have included many species that are native to North America, but have been translocated to states or regions in which they did not occur historically. We have also included some species that have expanded their historic range in recent decades because, in many cases, this range expansion has been facilitated, at least in part, by the activities and land uses of humans. The list may actually be considerably longer, but for many species, we cannot yet be sure that they are well established and will sustain themselves over time. For example, Florida agencies use a "10 year rule" of documentation of breeding and establishment in several counties before the species is put on its listed of introduced and established species. It appears that a large portion of the introduced vertebrates occur in Florida, Texas, California, and Hawaii; however, all states have a number of well-established introductions.

At least 81 species of introduced/invasive mammals have become established in parts of the US (Table 1). Mammals were mainly introduced for sport hunting, but also for food and fur (Kraus 2003). The largest single group is ungulates with 33 species. This group includes many species used for sport hunting (e.g., aoudad, gemsbok, nilgai, eland), but also feral populations of species that were used for work (e.g., horses, burros) or for food (e.g., cattle, pigs). The second largest group of mammals is rodents (18), many of which were introduced accidentally via cargo and transport vehicles (commensal rats and mice), but some were purposefully introduced for fur (e.g., nutria). Numerous carnivores (14 species) have been introduced, in some cases for their fur (e.g., foxes, raccoons), but also in efforts to control pests such as rats and snakes (e.g., mongoose, weasels). There are also large populations of feral, formerly companion animals (e.g., dogs, cats) throughout the US Interestingly, at least 6 species of primates have become established in parts of the US

At least 94 species of introduced/invasive birds have become established in the US (Table 2). Most introductions were as pets, but many were introduced for sport hunting (Kraus 2003). Most of these are passerines (39 species), but many are psitticines (22 species, popular animals in the pet industry). There are also a large number (20 species) of "upland game"/galliform birds (both native and non-native) that have been introduced to various parts of the US. Interestingly, many more avian species have been introduced to Hawaii than the mainland (Lever 1987).

At least 86 species of introduced/invasive herptiles have been introduced to the US (Table 3). Most introductions were as pets, but many were also introduced as accidentals in cargo (Kraus 2003). This group of animals is also very popular in the pet industry. Florida is a very large importer of herptiles which may be why that state seems to have the largest number of established species. The largest single group of herptiles is lizards (61 species). Other groups (frogs and toads, turtles, snakes) comprise smaller (6-9 species) and relatively similar numbers of species per group. Only one crocodilian (caiman) has become established in a few parts of the US.

**Table 1.** Mammals introduced into parts of the US.

**MARSUPIALS:**
Opossum (*Didelphis marsupialis*)
Brush-tailed rock wallaby (*Petrogale penicillata*)

**PRIMATES:**
Squirrel monkey (*Saimiri sciureus*)
Vervet monkey (*cercopithecus aethiops*)
Crab-eating Monkey (*Macaca fascicularis*)
Japanese macaque (*M. fuscata*)
Rhesus monkey (*M. mulatto*)
Chimpanzee (*Pan troglodytes*)

**INSECTIVORES:**
Nine-banded armadillo (*Dasypus novemcinctus*)
Pallas's mastiff bat (*Molossus molossus*)

**LAGOMORPHS:**
Eastern cottontail (*Sylvilagus floridanus*)
European rabbit (*Oryctolagus cuniculus*)
Snowshoe hare (*Lepus americanus*)
Black-tailed jackrabbit (*L. californicus*)
European hare (*L. europaeus*)
White-tailed jackrabbit (*L. townsendii*)

**RODENTS:**
Arctic ground squirrel (*Spermophilus parryii*)
Prairie dog (*Cynomys ludovicianus*)
Abert's squirrel (*Sciurus aberti*)
Mexican red-bellied squirrel (*S. aureogaster*)
Gray squirrel (*S.* carolin)nsis
Fox squirrel (*S. niger*)
Red squirrel (*S. vulgaris*)
Kangaroo rat (*Dipodomys ordii*)
Deer mouse (*Peromyscus maniculatus*)
Red-backed vole (*Clethrionomys rutilus*)
Muskrat (*Ondatra zibethicus*)
Polynesian rat (kiore) (*Rattus exulans*)
Norway (brown) rat (*R. norvegicus*)
Ship (black, roof) rat (*R. rattus*)
Gambian giant pouched rat (*Cricetomys gambianus*)
House mouse (*Mus musculus*)
Beaver (*Castor canadensis*)
Nutria (*Myocastor coypus*)

**CARNIVORES:**
Red fox (*Vulpes vulpes*)
Arctic fox (*Alopex lagopus*)
Feral dog (*Canis familiaris*)
Coyote (*C. latrans*)
Raccoon (*Procyon lotor*)
Coatimundi (*Nasua nasua*)
White-nosed coati (*N. narica*)
Stoat (ermine, short-tailed weasel) (*Mustela erminea*)
Least weasel (*M. nivalis*)
European polecat (*M. putorius*)
American mink (*M. vision*)
Small Indian mongoose (*Herpestes auropunctatus*)
Feral cat (*Felis catus*)
Jaguarundi (*F. yagouaroundi*)

**UNGULATES:**
Donkey (burro)(*Equus asinus*)
Feral horse (*E. caballus*)
Burchell's zebra (*E. burchelli*)
Feral pig (*Sus scrofa*)
Camel (*Camelus bactrianus*)
Axis deer (*Cervus axis*)
Fallow deer (*C. dama*)
Swamp deer (*C. duvauceli*)
Wapiti (American elk, red deer) (*C. elaphus*)
Sika deer (*C. Nippon*)
Sambar deer (*C. unicolor*)
Black-tailed deer (*Odocoileus hemionus*)
Roe deer (*Capreolus capreolus*)
Moose (*Alces alces*)
Reindeer (caribou)(*Rangifer tarandus*)
Pronghorn antelope (*Antilocapra Americana*)
Eland (*Taurotragus oryx*)
Nilgai (*Boselaphus tragocamelus*)
Water buffalo (*Bubalus bubalis*)
Feral cattle (*Bos Taurus*)
Bison (*Bison bison*)
Gemsbok (*Oryx gazelle*)
Blackbuck (*Antilope cervicapra*)
Mountain goat (*Oreamnos americanus*)
Chamois (*Rupicapra rupicapra*)
Musk-ox (*Ovibos moshatus*)
Himalayan tahr (*Hemitragus jemlahicus*)
Feral goat (*Capra hircus*)
Alpine ibex (*C. ibex*)
Aoudad (Barbary sheep) (*Ammotragus lervia*)
Mouflon sheep (*Ovis ammon*)
Feral sheep (*O. aries*)
Bighorn sheep (*O. canadensis*)

**Table 2.** Birds introduced into parts of the US.

**ANSERIFORMES:**
Mute swan (*Cygnus olor*)
Mandarin duck (*Aix galericulata*)
Muscovy duck (*Cairina moschata*)
American black duck (*Anas rubripes*)
Mallard (*A. platyrhynchos*)

**CICONIIFORMES:**
Cattle egret (*Bubulcus ibis*)

**GALLIFORMES:**
Plain chachalaca (*Ortalis vetula*)
Bobwhite quail (*Colinus virginianus*)
Mountain quail (*Oreortyx picta*)
California quail (*Lophortyx californicus*)
Gambel's quail (*L. gambellii*)
Scaled quail (*Callipepla squamata*)
Chukar partridge (*Alectoris chukar*)
Barbary partridge (*A. Barbara*)
Grey partridge (*Perdix perdix*)
Black francolin (*Francolinus francolinus*)
Grey francolin ( *F. pondicerianus*)
Erkel's francolin (*F. erkelii*)
Himalayan snowcock (*Tetragallus himalayensis*)
Common quail (coturnix quail)(*Coturnix coturnix*)
Chinese bamboo partridge (*Bambusicola thoracica*)
Kalij pheasant (*Lophura leucomelana*)
Red jungle fowl (*Gallus gallus*)
Common (ring-necked) pheasant (*Phasianus colchicus*)
Reeve's pheasant (*Syrmaticus reevesii*)
Helmeted guineafowl (*Numida meleagris*)
Common peafowl (*Pavo cristatus*)
White-tailed ptarmigan (*Lagopus leucurus*)
Common turkey (*Melagris gallopavo*)
Chesnut-bellied sandgrouse (*Pterocles exustus*)
Ruffed grouse (*Bonasa umbellus*)

**COLUMBIFORMES:**
Feral pigeon (rock dove) (*Columba livia*)
African collared dove (*Streptopelia roseogrisea*)
Eurasian collared dove (*S. decaocto*)
Spotted dove (*S. chinensis*)
Barred (zebra) dove (*Geopelia striata*)
White-winged dove (*Zenaida asiatica*)
Mourning dove (*Z. macroura*)

**PSITTACIFORMES:**
Monk parakeet (*Myiopsitta monachus*)
Ring-necked (rose-winged) parakeet (*Psittacula krameri*)
Canary-winged (white-winged) parakeet (*Brotogeris versicolurus*)
Blue-crowned parakeet (*Aratinga acuticaudata*)
Budgerigar (*Melopsittacus undulatus*)
Green-cheeked Amazon (red-crowned parrot) (*Amazona viridigenalis*)
Yellow-crowned Amazon (*A. ochrocephula*)
Yellow-headed Amazon (*A. oratrix*)
Blue-fronted Amazon (*A. aestiva*)
Lilac-crowned parrot
Senegal parrot (*Poicephalus senegalus*)
Blue-fronted conure (*Aratinga cruentata*)
Brown-throated conure (*A. pertinax*)
Mitered conure (*A. mitrata*)
Nanday conure (black-hooded parakeet) (*Nandayus nenday*)
Lovebird (*Agapornis* spp.)
Cockatiel (*Nymphicus hollandicus*)
Sulphur-crested cockatoo (*Cacatua galerita*)
White-crested cockatoo (*C. alba*)
Giffin's cockatoo (*Cacatua goffini*)

**STRIGIFORMES:**
Barn owl (*Tyto alba*)

**PASSERIFORMES:**
Edible-nest swiftlet (*Aerodramus fuciphagus*)
Mariana swiftlet (*A. bartschi*)
Skylark (*Alauda arvensis*)
Western meadowlark (*Sturnella neglecta*)
Red-vented bulbul (*Pycnonotus cafer*i)
Red-whiskered bulbul (*P. jocosus*)
Northern mockingbird (*Mimus polyglottos*)
White-rumped shama (*Copsychus malabaricus*)
Melodious laughing thrush (*Garrulax canorus*)
Greater necklaced laughing thrush (*G. pectoralis*)
Red-billed leiothrix (*Leiothrix lutea*)
Japanese bush warbler (*Cettia diphone*)
Varied tit (*Parus varius*)
Japanese white-eye (*Zosterops japonica*)
Saffron finch (*Sicalis flaveola*)
Yellow-faced grassquit (*Tiaris olivacea*)
Red-crested cardinal (*Paroaria coronata*)
Yellow-billed cardinal (*P. capitata*)
Common (northern ) cardinal (*Cardinalis cardinalis*)
Spot-breasted oriole (*Icterus pectoralis*)
Yellow-fronted canary (*Serinus mozambicus*)
Common canary (*S. canaria*)
House finch (*Carpodacus mexicanus*)
Red-cheeked condon-bleu (*Uraeginthus benglaus*)
Lavender waxbill (*Estrilda caerulescens*)
Orange-cheeked waxbill (*E. melpoda*)
Common waxbill (*E. astrild*)
Black-rumped waxbill (*E. troglodytes*)
Red avadavat (*Amandava amandava*)
Nutneg manikin (*Lonchura punctulata*)
Black-headed manikin (*L.malacca*)
Warbling silverbill (*L. malabarica*)
Orange bishop (*Euplectes franciscanus*)
Java sparrow (*Padda oryzivora*)
House sparrow (*Passer domesticus*)
European tree sparrow (*P. montanus*)
European starling (*Sturnus vulgaris*)
Common mynah (*Acridotheres tristis*)
Hill mynah (*Gracula religiosa*)
Crested mynah (*Acridotheres cristatellus*)

**Table 3.** Reptiles and amphibians introduced into parts of the US.

**FROGS AND TOADS:**
Giant toad (cane toad)(*Bufo marinus*)
Green-and-black poison dart frog (*Dendrobates auratus*)
Greenhouse frog (*Eleutherodactylus planirostris*)
Coqui frog (*E. coqui*)
Cuban treefrog (*Osteopilus septentrionalis*)
Rio Grande leopard frog (*Rana berlandieri*)
Green frog (*R. clamitans*)
American bullfrog (*R. catesbeiana*)
Japanese wrinkled frog (*Glandirana rugosa*)
African clawed frog (*Xenopus laevis*)

**SALAMANDERS:**
Tiger salamander (*Ambystoma tigrinum*)

**TURTLES:**
Red-eared slider (*Trachemys scripta elegans*)
Painted turtle (*Chrysemys picta*)
Spiny softshell (*Trionyx spiniferus*)
Wattle-necked softshell (*Palea steindachneri*)
Chinese softshell (*Pelodiscus sinensis*)
Snapping turtle (*Chelydra serpentine*)

**CROCODILIANS:**
Spectacled (common) caiman (*Caiman crocodilus*)

**LIZARDS:**
Red-headed agama (*Agama agama*)
Giant ameiva (*Ameiva ameiva*)
Large-headed anole (*Anolis cybotes*)
Green anole (*A. carolinensis*)
Bark anole (*A. distichus*)
Hispaniolan green anole (*A. chlorocyanus*)
Puerto Rican crested anole (*A. cristatellus*)
Knight anole (*A. equestris*)
Cuban green anole (*A. porcatus*)
Jamaican giant anole (*A. garmani*)
Marie Gallant Sail-tailed anole (*A. ferreus*)
Brown anole (*A. sagrei*)
Brown basilisk (*Basiliscus vittatus*)
Veiled chameleon (*Chamaeleo calyptratus*)
Jackson's chameleon (*C. jacksonii*)
Butterfly lizard (*Leiolepis belliana*)
Oriental garden lizard (variable bloodsucker) (*Calotes versicolor*)
Rainbow whiptail lizard (*Cnemidophorus lemniscatus*)
Giant whiptail lizard (*C. [Aspidoselis] motaguae*)
New Mexico whiptail lizard (*C. neomexicanus*)
Plateau striped whiptail (*C. velox*)
Northern curlytail lizard (*Leiocephalus carinatus*)
Red-sided curlytail lizard (*L. schreibersii*)
Common wall lizard (*Podarcis muralis*)
Italian wall lizard (*P. sicula*)
Texas horned lizard (*Phrynosoma cornutum*)
Black (Gray's) spinytail iguana (*Ctenosaura similis*)
Mexican spinytail iguana (*C. pectinata*)
Green iguana (*Iguana iguana*)
Ashy gecko (*Sphaerodactylus elegans*)
Ocellated gecko (*S. argus*)
Gold dust day gecko (*Phelsuma laticauda*)
Giant day gecko (*P. madagascariensis*)
Orange-spotted day gecko (*P. guimbeaui*)
Moorish gecko (*Tarentola mauritanica*)
Ringed wall gecko (*T. annularis*)
Mourning gecko (*Lepidodactylus lugubris*)
Multilating gecko (*Gehyra mutilata*)
Rough-tailed gecko (*Cyrtopodion scabrum*)
Tokay gecko (*Gekko gecko*)
Common house gecko (*Hemidactylus frenatus*)
Tropical house gecko (wood stave) (*H. mabouia*)
Asian flat-tailed gecko (*H. platyurus*)
Indo-Pacific gecko (*H. garnotti*)
Mediterranean gecko (*H. turcicus*)
Yellow-headed gecko (*Gonatodes albogularis*)
Indo-Pacific tree gecko (*Hemiphyllodactylus typus*)
Moth skink (*Lipinia noctua*)
Azure-tailed skink (*Emoia impar*)
Copper-tailed skink (*E. cyanura*)
Many-lined grass skink (*Mabuya multifasciata*)
Pacific snake-eyed skink (*Cryptoblepharus poecilopleurus*)
Plague skink (*Lampropholis delicate*)
Western green lacerta (*Lacerta bilineata*)
Nile monitor (*Varanus niloticus*)
Argentina giant tegu (*Tupinambis merianae*)

**SNAKES:**
Common boa (*Boa constrictor*)
Burmese python (*Python molurus*)
Brahminy blind snake (*Ramphotyphlops braminus*)
Javan filesnake (*Acrochordus javanicus*)
Diamondback water snake (*Nerodia rhombifer*)
Brown treesnake (*Boiga irregularis*)

Although we do not address introduced/invasive fish in detail in this paper, it is important to acknowledge the substantial size of this group: at least 150 introduced species (Fuller et al. 1999, Fuller 2003). While some of these were accidental introductions (e.g., from ballast water), most were for sport fishing (e.g., trout [*Oncorhynchus* spp., *Salmo* spp., *Salvelinus* spp.], bass [*Micropterus* spp.]), but many were pet or bait fish releases (Fuller 2003). A few were introduced as a food source (e.g., tilapia [*Tilapia* spp.]) and a few species (e.g., grass carp [*Ctenopharyngodon idella*], mosquito fish [*Gambusia affinis*]) were widely introduced to control aquatic vegetation or mosquito larvae. More details on fish introductions can be found in the reviews by Fuller (2003), Fuller et al. (1999), Pimentel et al. (2005), and Pitt and Witmer (2007).

## EXAMPLES OF PROBLEMATIC INTRODUCED VERTEBRATES IN THE UNITED STATES

A number of species within each major taxonomic group of vertebrates pose serious problems over portions of the US. We give a few examples in each group, based on one or more of these criteria: their widespread nature and population sizes, the seriousness of the problems they cause, the amount of investment in prevention and control, and the number of requests of USDA/APHIS Wildlife Services (WS) to deal with specific damage situations. WS involvement in invasive vertebrate damage situations was compiled and discussed by Bergman et al. (2002) and Rennie et al. (2004).

Feral cats are found throughout the US and cause significant predation of native birds and other native animals (Pimentel et al 2005, Witmer et al. 2005, Pitt and Witmer 2007). Feral dogs can be found in most of the states and territories of the US. They pose human safety issues, prey on livestock, and hybridize with some species of native canids (Pimentel et al. 2005, Witmer et al 2005). Feral pigs are found in at least half of the states in the US. They cause serious environmental degradation, prey on native species, damage crops, and pose a disease hazard to livestock and wildlife (Witmer et al. 2003, Pimentel et al. 2005, Pitt and Witmer 2007). Several species of herbivores (exotic and feral rabbits and introduced nutria) also cause ecosystem and crop damage (Witmer and Lewis 2001). One or more species of commensal rats and mice occur everywhere worldwide and in all states and territories of the US. These rodents cause disease and sanitation problems, consumption and contamination of foodstuffs (both in the field and in storage), and property damage (Witmer et al. 1995, Pimentel et al. 2005, Pitt and Witmer 2007). They have also caused the extinction or endangerment of many endemic species on islands (Witmer and Eisemann 2005).

Starlings, pigeons, and house sparrows are found almost worldwide and throughout almost all of the US. They are so well established in the US as to be considered "naturalized" and many people no longer even consider them invasive species. Nonetheless, they cause sanitation and disease problems, compete with native birds, and consume and contaminate livestock feed (Witmer and Lewis 2001, Pimentel et al. 2005). Other serious invasive bird problems are more localized, such as mute swans in several northeastern states. They pose human safety concerns because of their aggressive behavior and they compete with native bird species (Avery and Tillman 2005). Populations of monk parakeets have become established in several states where they cause power outages by nesting in transformers (Avery and Tillman 2005). They also pose a significant threat of crop damage if populations become sizable in agricultural areas. Finally, ring-necked pheasants were introduced to many states for sport hunting. They cause serious crop damage in some localized situations and may compete for resources with native upland bird species (Witmer and Lewis 2001).

Perhaps the most widespread invasive herptile in the US is the bullfrog. While native to the eastern US, bullfrogs have been introduced to many western states. They prey on many aquatic animal species across all taxa, compete for resources, and have contributed to the threatened or endangered status of many regionally-endemic species of frogs (Witmer and Lewis 2001, Pitt et al. 2005, Pitt and Witmer 2007). Other invasive herptile problems in the US are much more localized. In Guam, the brown treesnake predates upon, and competes with, native species of vertebrates and has caused the extinction of several of those species. They also regularly cause power outages and pose a safety hazard to people, especially children (Pimentel et al. 2005, Pitt et al. 2005, Pitt and Witmer 2007). Coqui frogs have become well established in Hawaii where their calling all night disturbs peoples' rest and has caused a decline in property values (Pitt et al. 2005, Pitt and Witmer 2007). Several large, aggressive, carnivorous species of herptiles (in particular, Burmese pythons and Nile

monitor lizards) have become established in parts of Florida. They pose human and companion animal safety hazards, as well as issues of competition and predation with native vertebrate species (Pitt and Witmer 2007).

## MANAGEMENT AND ERADICATION OF INTRODUCED/INVASIVE VERTEBRATES

A wide array of methods is used to manage invasive vertebrates in the US and these vary somewhat by taxonomic group (Table 4). Details on most of these methods, how they are used, and their advantages and disadvantages were presented by Conover (2002), Hygnstrom et al. (1994), and VerCauteren et al. (2005). Eradication strategies are more complex and are discussed by taxonomic group.

Management of invasive rodents most often utilizes rodenticides, and primarily anticoagulants (Howald et al. 2007). Traps (kill traps, live traps, glue boards) are used in some situations, but to a much lesser extent. These methods are supplemented in and around buildings, with practices of exclusion, sanitation, and habitat modification (Timm 1994). Day and night shooting is used with some larger species (e.g., nutria; LeBlanc 1994). Most island eradications have utilized anticoagulant rodenticides---hand broadcast, in bait stations, or aerially broadcast (Howald et al. 2007).

A wide array of methods is used for carnivore management and eradication (Nogales et al. 2003, Witmer et al. 2005). Carnivores are captured with live-traps, leg-hold traps, and snares. For smaller species, kill traps (e.g., conibear traps) are also used. Shooting (day, night, with calling) is often used. Occasionally, aerial shooting is used. Exclusion is sometimes used, especially to protect small colonies of endangered species. Toxicants are used on a limited basis: toxic baits and the M-44 cyanide device are sometimes used, especially on islands (Nogales et al. 2003, Witmer et al. 2005). A number of methods are not effective with carnivores and rodents (frightening devices, repellents, taste aversion), although research continues on these and other methods. Few carnivore eradications have been attempted in the US. However, introduced foxes have been eradicated from several Aleutian Islands with the use of shooting, traps, and toxic baits (Ebbert 2000).

A wide array of methods is used for ungulate management and eradication (Butchko et al. 2003, Campbell and Donlan 2005, Lowney et al. 2005). These include shooting (day, night, over bait, aerial), trapping (individual cage traps, snares, group/corral cage traps), pursuit with dogs, exclusion, food removal, and Judas animals. Eradications have occurred on a few small islands and on some sizable, fenced/contained areas of the mainland (Butchko et al. 2003, Campbell and Donlan 2005, Lowney et al. 2005). Generally, several methods have been employed to assure success.

**Table 4.** Methods[a] used to manage and eradicate invasive vertebrate species in the US.

| Method: / Group: | Trap | Shoot | Toxi-cants | Dogs | Judas animals | Introduced predator | Habitat Manip. | Barriers | Other methods |
|---|---|---|---|---|---|---|---|---|---|
| Rodents | X | N/O | X | N/O | | X | X | X | B |
| Ungulates | X | X | | X | X | | X | X | F |
| Carnivores | X | X | X | X | | X | | X | B, C, T |
| Birds | X | X | X | X | | | X | X | F, D |
| Herptiles | X | | X | X | | | | X | |

[a]N/O = nutria only, B = bounty, F = frightening devices, C = compensation, T = trap/neuter/release, D = egg/nest destruction

Methods used to manage invasive bird populations are similar to those used for mammals: traps (with or without live bird decoys), shooting, exclusion, and limited use of toxicants (DRC 1339; Witmer and Lewis 2001, Millet et al. 2004, Avery and Tillman 2005, Pitt and Witmer 2007). The exceptions are that egg and nest destruction is sometimes used with birds and frightening devices are often used to protect relatively small areas. While few, if any, bird eradications have been conducted in the US, Millett et al. (2004) used shooting and toxicants to eradicate invasive common mynahs from several small islands in the Seychelles. They noted that larger islands were much more difficult, if not impossible, to eradicate, and that re-invasion was often a problem on all but the most remote islands.

Our tool box for management and eradication of invasive vertebrates is perhaps weakest for amphibians and reptiles (Witmer and Lewis 2001, Pitt and Witmer 2007). Most methods development has occurred for brown treesnakes in Guam and coqui frogs in Hawaii (Pimentel et al. 2005, Pitt et al. 2005, Pitt and Witmer 2007). Trapping, hand-capture or pit fall traps are perhaps most commonly used with herptiles. Drift fences are often used to increase effectiveness by directing animals to traps or pit falls. Night search-and-capture with spotlights can be used, and with brown treesnakes, night fence searches are conducted. Detector dogs are used to inspect cargo for brown treesnakes and to help locate Burmese pythons in Florida's Everglades National Park. Toxicants have been developed and registered for brown treesnakes (acetaminophen) and for coqui frogs (citric acid and hydrated lime solutions; Pitt et al. 2005, Pitt and Witmer 2007). We are not aware of any eradications of introduced herptiles in the US.

## MAJOR ISSUES IN DEALING WITH INVASIVE VERTEBRATES IN THE UNITED STATES

While some progress has been, and is being, made with invasive vertebrates in the US, there are still many challenges and issues to resolve (National Invasive Species Council 2001, Pimentel et al. 2005). The major emphasis, in terms of attention and funding, for invasive species in the US has been focused on plants, insects, and pathogens (Pimentel et al. 2005). Relatively little effort and few resources have been directed to vertebrates with the main exceptions of brown treesnakes in Guam and feral pigs (in many states).

Public perception and lack of support have affected efforts to manage or eradicate vertebrate species in the US, as elsewhere in the world (National Invasive Species Council 2001). Knowledge levels regarding invasive species and the harm they can cause are relatively low amongst the general public (National Invasive Species Council 2001, Conover 2002). Furthermore, the public does not readily distinguish between native and non-native species: as long as an animal looks nice and is not threatening people or causing undue harm, the public tends to view species equally (Wittenberg and Cock 2001). Once it has been established that a species will not cause undo environmental or human resource harm, it can be placed on a "white" list (Fowler et al. 2007). By and large, in the US, species importations are viewed as "innocent until proven guilty" ("gray" list) and what is needed is the development of a much more inclusive prohibited species "black" list (Witmer and Lewis 2001, Fowler et al. 2007, Pitt and Witmer 2007). The ultimate solution to an invasive species is the eradication of all individuals in a given area; however, much of the public has a strong dislike for the killing of animals (Conover 2002). Certain species such as feral cats, feral dogs, wild horses, and primates are particularly sensitive species to address. Furthermore, much of the public has a strong fear and distrust of chemicals, and in particular, toxicants. Hence, the management of invasive vertebrates, like all wildlife, is being conducted in an increasingly complex arena (Conover 2002, Fall and Jackson 2002).

The pet industry is a well organized, large, and influential industry in the US (Ginsburg 2004). Exotic pets are very popular with a sizable portion of the public. And yet, the pet industry is a major pathway for the introduction of vertebrates into the US (Kraus 2003). Very few vertebrate species are prohibited from entry into the US with a prevailing attitude of "innocent until proven guilty" (Witmer and Lewis 2001, Pitt and Witmer 2007). Greater cooperation, enforcement, and regulation of the pet industry is needed.

Access to all relevant land and properties is essential for the successful management and eradication of invasive vertebrates. However, mangers often face the situation where the work is needed across a wide array of jurisdictions and ownerships. Getting permission to access all these

areas rarely occurs and can prevent the success of even a well-planned, well-funded eradication effort. Furthermore, land management mandates and regulations of federal and state agencies vary considerably. This affects the type of management activities (burning, chemical use), type of vehicles, and tools (leg-hold traps, firearms, toxicants) that can be used on certain properties. Some laws actually protect invasive vertebrate species, such as the Wild Horse and Burro Act and the Migratory Bird Treaty Act. The latter was recently amended to exclude some non-native bird species in the US such as the mute swan.

Finally, there is an overall lack of coordination and cooperation across jurisdictions and agencies of all levels of government in the US. One of the goals of the National Invasive Species Management Plan (National Invasive Species Council 2001) is to rectify that situation. Eradicating an invasive vertebrate species is rarely an easy undertaking. Very careful planning is needed, along with adequate resources, public and agency buy-in, highly trained and motivated personnel, contingency plans, and a sustained effort (Broome 2005). Each situation is unique in one or more ways; hence, a cook-book approach cannot be used (Broome 2005).

With the possible exception of rodents and ungulates, the methods and strategies used for management, and especially eradication, of invasive vertebrates need improvement (Wittenberg and Cock 2001). Much research needs to be conducted to improve detection methods. Attractants are needed to attract individuals to traps, bait stations, and detection stations. Method improvements are needed to ensure effective and safe delivery of toxicants, vaccines, and fertility control agents. Trained, rapid response teams are needed for many more invasive species. Accessible databases on potential invasive species are needed to summarize species identification, biology, ecology, and effective detection and management methods. The databases should also identify expertise and literature that can be consulted. Although a variety of databases and websites exist (Sellers et al. 2005), it would be very useful if these could be centralized and standardized (Sellers et al. 2004). Finally, risk assessments are needed to determine on which species we should focus our efforts and resources (Hayes 2003).

## CONCLUSION

At least 161 species of introduced/invasive vertebrates occur in the US. We suspect that invasive vertebrate species will continue to challenge land and resource managers, ecologists, and biologists for a long time to come. We also suspect that the list of invasive vertebrate species in the US will continue to grow; but, hopefully, some species will also be removed from the list. We have had some good successes with invasive species management and eradications, especially on islands, but also on some areas of the mainland. As a result of this, along with our collaborations with international colleagues and a growing interest and involvement by the public and agencies, we are becoming more knowledgeable and pro-active in responding to invasive vertebrate species. We still have a long way to go in terms of national organization and cooperation on these issues, resolving various logistical and financial issues, and improving methods and strategies for many more species.

## LITERATURE CITED


American Ornithologist's Union. 1998. Check-list of North American birds. American Ornithologists' Union, Washington, D.C., USA.

Avery, M. and E. Tillman. 2005. Alien birds in North America—challenges for wildlife managers. Proceedings of the Wildlife Damage Management Conference 11:82-89.

Bergman, D., M. Chandler, and A. Locklear. 2002. The economic impact of invasive species to wildlife services' cooperators. Pages 169-178 *in* L. Clark, editor. Human conflicts with wildlife: economic considerations. USDA National Wildlife Reseaerch Center, Fort Collins, Colorado, USA.

Broome, K. 2005. Rat eradication—how to get it right without a recipe. Proceedings of the Australasian Vertebrate Pest Conference 13:152-157.

Bury, R., and R. Luchenbach. 1976. Introduced amphibians and reptiles in California. Biological Conservation 10:1-14.

Butchko, P., A. Duffiney, and A. Montoney. 2003. Eradication of a bovine tuberculosis-postive captive cervid herd in northeast Michigan. Proceedings of the Wildlife Damage Management Conference 10:408-413.

Campbell, K., and J. Donlan. 2005. Feral goat eradication on islands. Conservation Biology 19:1362-1374.

CONOVER, M. 2002. Resolving human-wildlife conflicts. Lewis Publishers, Boca Raton, Florida, USA.

EBBERT, S. 2000. Successful eradication of introduced arctic foxes from large Aleutian islands. Proceedings of the Vertebrate Pest Conference 19:127-132.

FALL, M., AND W. JACKSON. 2002. The tools and techniques of wildlife damage management—changing needs: an introduction. International Biodeterioration and Biodegradation 49:87-91.

FOWLER, A., D. LODGE, AND J. HSIA. 2007. Failure of the Lacey Act to protect U.S. ecosystems against animal invasions. Frontiers in Ecology and the Environment 5:353-359.

FULLER, P. 2003. Freshwater aquatic vertebrate introductions in the United States: patterns and pathways. Pages 123-151 *in* G. Ruiz and J. Carlton, editors. Invasive species: vectors and management strategies. Island Press, Washington, D.C., USA.

FULLER, P., L. NICO, AND J. WILLIAMS. 1999. Nonindigenous fishes introduced into inland waters of the United States. American Fisheries Society Special Publication 27. American Fisheries Society, Bethesda, Maryland, USA.

GINSBURG, J. 2004. Dinner, pets, and plagues by the bucketful. The Scientist 18:28-29.

HAWAII AUDUBON SOCIETY. 1989. Hawaaii's birds. Hawaii Audubon Society, Honolulu, Hawaii, USA.

HAYES, K. 2003. Biosecurity and the role of risk assessment. Pages 382-414 *in* G. Ruiz and J. Carlton, editors. Invasive species: vectors and management strategies. Island Press, Washington, D.C., USA.

HOWALD, G., C. DONLAN, J. GALVAN, J. RUSSELL, J. PARKES, A. SAMANIEGO, Y. WANG, D. VEITCH, P. GENOVESI, M. PASCAL, A. SAUNDERS, AND B. TERSHY. 2007. Rodent eradications on islands. Conservation Biology in press.

HYGNSTROM, S., R. TIMM, AND G. LARSON. 1994. Prevention and control of wildlife damage. University of Nebraska Cooperative Extension Service, Lincoln, Nebraska, USA.

KRAUS, F. 2003. Invasion pathways for terrestrial vertebrates. Pages 68-92 *in* G. Ruiz and J. Carlton, editors. Invasive species: vectors and management strategies. Island Press, Washington, D.C., USA.

KRAUS, F. In press. Alien species. *In* B. Crother, editor. Scientific and standard English names of amphibians and reptiles of North America north of Mexico, with comments regarding confidence in our understanding. Society for the Study of Amphibians and Reptiles (SSAR) Herpetological Circular. Department of Biological Sciences, Southeastern Louisiana University, Hammond, Louisiana, USA.

LEBLANC, D. 1994. Nutria. Pages B-7 - B-80 *in* S. Hygnstrom, R. Timm, and G. Larson, editors. Prevention and Control of Wildlife Damage. University of Nebraska Cooperative Extension Service, Lincoln, Nebraska, USA.

LEVER, C. 1987. Naturalized birds of the world. John Wiley & Sons, New York, USA.

LONG, J. 1981. Introduced birds of the world. Universe Books, New York, USA.

LONG, J. 2003. Introduced mammals of the world. CSIRO Publishing, Collingwood, Australia.

LOWNEY, M., P. SCHOENFELD, W. HAGLAN, AND G. WITMER. 2005. Overview of impacts of feral and introduced ungulates on the environment in the eastern Untied States and Caribbean. Proceedings of the Wildlife Damage Management Conference 11:64-81.

MESHAKA, W. 2006. An update on the list of Florida's exotic amphibians and reptile species. Journal of Kansas Herpetology 19:16-17.

MILLETT, J., G. CLIMO, AND N. SHAH. 2004. Eradication of common mynah populations in the granitic Seychelles: successes, failures and lessons learned. Advances in Vertebrate Pest Management 3:169-183.

MOONEY, H., AND J. DRAKE. 1986. The invasions of plants and animals into California. Pages 250-272 *in* H. Mooney and J. Drake, editors. Ecology and biological invasions of North America and Hawaii. Springer-Verlag, New York, USA.

MOULTON, M. AND S. PIMM. 1986. Species introductions to Hawaii. Pages 231-249 *in* H. Mooney and J. Drake, editors. Ecology and biological invasions of North America and Hawaii. Springer-Verlag, New York, USA.

NATIONAL INVASIVE SPECIES COUNCIL. 2001. Meeting the invasive species challenge: management plan. National Invasive Species Council, Washington, D.C., USA.

NOGALES, M., A. MARTIN, B. TERSHY, C. DONLAN, D. VEITCH, N. PUERTA, B. WOOD, AND J. ALONSO. 2003. A review of feral cat eradication on islands. Conservation Biology 18:310-319.

PIMENTEL, D., R. ZUNIGA, AND D. MORRISON. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52:273-288.

PITT, W., D. VICE, AND M. PITZLER. 2005. Challenges of invasive reptiles and amphibians. Proceedings of the Wildlife Damage Management Conference 11:112-119.

PITT, W., AND G. WITMER. 2007. Invasive predators: a synthesis of the past, present, and future. Pages 265-293 *in* A. Elewa, editor. Predation in organisms—a distinct phenomenon. Spring-Verlag, Heidelberg, Germany.

RENNIE, T., R. FRANCIS, L. HENZE, AND W. PITT. 2004. Wildlife Services concept plan for invasive vertebrate species. USDA/APHIS Wildlife Services, Washington, D.C., USA.

SELLERS, E., R. MUIR, AND A. SIMPSON. 2004. Summary report on the experts meeting on

implementation of a global invasive species information network. National Biological Information Infrastructure, Information International Associates, Inc., Oak Ridge, Tennessee, USA.

SELLERS, E., A. SIMPSON, AND S. CURD-HETRICK. 2005. List of invasive alien species (AIS) online databases and databases containing IAS information. National Biological Information Infrastructure, Information International Associates, Inc., Oak Ridge, Tennessee, USA.

TEER, J. 2003. Nonnative large mammals in North America. Pages 1180-1187 *in* G. Feldhamer, B. Thompson, and J. Chapman, editors. Wild mammals of North America. 2nd Editon. The Johns Hopkins University Press, Baltimore, Maryland, USA.

TIMM, R. 1994. Norway rats. Pages B-105 – B-120 *in* S. Hygnstrom, R. Timm, and G. Larson, editors. Prevention and control of wildlife damage. University of Nebraska Cooperative Extension Service, Lincoln, Nebraska, USA.

VEITCH, C., AND M. CLOUT. 2002. Turning the tide: the eradication of invasive species. Occasional paper of the IUCN Species Survival Commission No. 27. World Conservation Union (IUCN), Gland, Switzerland.

VERCAUTEREN, K., R. DOLBEER, AND E. GEESE. 2005. Identification and management of wildlife damage. Pages 740-778 *in* C. Braun, editor. Techniques for wildlife investigations and management. The Wildlife Society, Bethesda, Maryland, USA.

WITMER, G., M. FALL, AND L. FIEDLER. 1995. Rodent control, research needs, and technology transfer. Pages 693-697 *in* J. Bissonette, editor. Integrating people and wildlife for a sustainable future. The Wildlife Society, Bethesda, Maryland, USA.

WITMER, G., AND J. LEWIS. 2001. Introduced wildlife in Oregon and Washington. Pages 423-443 *in* D. Johnson and T. O'Neil, editors. Wildlife-habitat relationships in Oregon and Washington. Oregon State University Press, Corvallis, Oregon, USA.

WITMER, G., R. SAUNDERS, AND A. TAFT. 2003. Feral swine—are they a disease threat to livestock in the United States? Proceedings of the Wildlife Damage Management Conference 10:316-325.

WITMER, G., B. CONSTANTIN, AND F. BOYD. 2005. Feral and introduced carnivores: issues and challenges. Proceedings of the Wildlife Damage Management Conference 11:90-101.

WITMER, G., AND J. EISEMANN. 2005. An overview of the 2nd national invasive rodent summit. Proceedings of the Wildlife Damage Management Conference 11:102-111.

WITTENBERG, R. AND M. COCK. 2001. Invasive alien species: a toolkit of best prevention and management practices. CAB International, Wallingford, United Kingdom

# CLIMATE MATCHING AS A TOOL FOR PREDICTING POTENTIAL NORTH AMERICAN SPREAD OF BROWN TREESNAKES

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***Abstract:*** Climate matching identifies extralimital destinations that could be colonized by a potential invasive species on the basis of similarity to climates found in the species' native range. Climate is a proxy for the factors that determine whether a population will reproduce enough to offset mortality. Previous climate matching models (e.g., Genetic Algorithm for Rule-set Prediction [GARP]) for brown treesnakes (*Boiga irregularis)* were unsatisfactory, perhaps because the models failed to allow different combinations of climate attributes to influence a species' range limits in different parts of the range. Therefore, we explored the climate space described by bivariate parameters of native range temperature and rainfall, allowing up to two months of aestivation in the warmer portions of the range, or four months of hibernation in temperate climes. We found colonization area to be minimally sensitive to assumptions regarding hibernation temperature thresholds. Although brown treesnakes appear to be limited by dry weather in the interior of Australia, aridity rarely limits potential distribution in most of the world. Potential colonization area in North America is limited primarily by cold. Climatically suitable portions of the United States (US) mainland include the Central Valley of California, mesic patches in the Southwest, and the southeastern coastal plain from Texas to Virginia.

***Key Words:*** *Boiga irregularis*, brown treesnake, climate match, GARP, invasive species, native range, United States.

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## INTRODUCTION

The brown treesnake (*Boiga irregularis*) is native to northern and eastern Australia, eastern Indonesia, Papua New Guinea, and most of the Solomon Islands. Following World War II, the brown treesnake was accidentally translocated in cargo to the formerly snake-free island of Guam, with devastating consequences for native birds, bats, and lizards (Savidge 1987, Fritts and Rodda 1998). The US Department of Agriculture (USDA) Wildlife Services (WS) now conducts a cargo inspection program to keep the snake from spreading from Guam to other areas.

In invasive species management, climate matching is used primarily for two purposes: to assess overall invasion risk and prioritize destination-specific management actions. An example of the former is Australia's algorithm for assessing the risk of importing exotic reptile species (Bomford et al. 2005), which attributes a higher risk to species that could colonize a larger proportion of the destination country. An example of the latter is WS's risk management of cargo outbound from the island of Guam (D. S. Vice, personal communication), which concentrates interdiction effort (dog-aided searches, etc.) on shipments bound for destinations most likely to support colonization by brown treesnakes. To date, the climatic division of destinations into high- and low-risk categories has been qualitative. While it is clear that Honolulu is at greater risk than Anchorage, uncertainty surrounds intermediate climates such as that experienced by Los Angeles.

A vast literature exists on climate matching not only for the purposes cited above, but also for predicting species occurrences within unsurveyed native range sites (Scott et al. 2002), and projecting range shifts in response to global climate change (Araujo and New 2007). Within the invasive species field, a continuum of approaches is recognizable, one end of which emphasizes the focal species' physiological tolerances as determined in the lab and the other end relying on automated searching for statistical correlations between site occupancy and climate values. An example of the former is the software package CLIMEX (and its predecessors, including BIOCLIM), which has been used extensively for

insects and plants, but also terrestrial vertebrates (Nix 1986, Pheloung 1996, Sutherst et al. 1996, Elith et al. 2006). Mapped range limits are associated with climate values, but the program also seeks physiological data of relevance such as minimum temperature tolerated, diapause duration, temperature threshold for emergence, and so forth. CLIMEX has default climate-matching algorithms that it will use to predict limiting conditions based on broad ecological categories that the investigator assigns (e.g., tropical versus temperate setting).

An example of the latter are programs such as Genetic Algorithm for Rule-set Prediction (GARP) and WhereWhy (Stockwell and Noble 1992, Stockwell 1999, Stockwell and Peters 1999, Stockwell et al. 2005), which automate the exploration of statistical association between occupied sites and climate values (other types of values could be included, for example soil type, but those geographic information system [GIS] layers must be provided by the user). GARP is genetic in the sense that in each generation (of the model) a climate variable (e.g., mean maximum July temperature) is randomly added to the model, as if it were a genetic mutation. The program then compares the goodness of model fit of the descendant model to that of the progenitor and retains the "mutation" if the fit improves. This process continues iteratively until no further improvements are obtained from a specified number of mutations. Like evolution, the mutations are not directed, so there is no guarantee that the resultant model will be optimal. Furthermore, although climate variables are inherently relevant to identifying range limits, the specific variables are chosen on statistical rather than biological grounds. In a statistical sense, the GARP algorithm is "fishing," with attendant risks of identifying a correlated but non-causal link. The successor program to GARP is WhereWhy, which adds some look-ahead capability so that the better models are discovered more expeditiously (Stockwell et al. 2005). Nevertheless, GARP and WhereWhy produce a large number of models whose outcomes are tabulated spatially to provide the proportion of colonization "successes" for each map pixel.

These programs have been criticized on a number of grounds (O'Connor 2002, Guisan et al. 2006), including lack of statistical protection against fishing and overfitting, lack of independence in the source data set, lack of independence between fitting and testing data sets, climatic irrelevance of localities that are in the interior of a species' range, assumption that a single suite of climate variables is responsible for diverse range limits, and the assumption that climate as a proxy for niche will apply to a novel environment in the same manner as it applies in the native range (Broennimann et al. 2007).

These problems are evident in the GARP map prepared by Stockwell et al. (2005) for the brown treesnake. The source data for that work were Australian National Museum specimen localities. Overfitting was evident in the highest probability of colonization assigned to the two sites (vicinities of Sydney and Brisbane) from which most of the specimens were obtained. Occupied but infrequently-collected sites midway between Sydney and Brisbane, or north of Brisbane were scored as low probability of colonization (the same rating as improbable sites such as Tasmania, Colorado or northern China). Because the same data set (though different points) were used for both training and testing the GARP algorithm, Stockwell et al. (2005) claimed an accuracy score of 80%, even though the GARP models failed to identify most of the Australian native range as having high probability of colonization.

The assumption that climate delineates niche and does so in the same way in native and introduced ranges is difficult to fully test. Ideally, one would like to have a prediction well documented prior to a colonization, and retrospectively compare the equilibrium distribution of the colonized area to that which was *a priori* predicted. Brown treesnake colonization of North America has not yet occurred, fortunately, but if it does, the GARP map produced by Stockwell et al. (2005) will provide a useful basis for comparison. In our opinion it is notable that the highest probability of colonization was not associated with southern Florida (which we judge to be highly vulnerable), but with the arid southern part of Texas. Southern Florida was judged much less vulnerable than northern Florida, and all of the Northeastern and upper Midwestern states (to the map's truncation at 40°N) were mapped in the same risk class as peninsular Florida and most of the occupied parts of Australia. We believe that this erroneous assessment is based on GARP choosing variability in temperature (rather than temperature) as the key association with the climatically-variable areas from which most of the Australian National Museum specimens were collected (notably the Blue Mountains west of Sydney). This illustrates a hazard of automated searches for climatic correlates.

We preferred to investigate the brown treesnake's climate envelope by starting with biological first principles. We recognized that climate is a proxy, and may be a poor proxy, for the ecological conditions that permit a population to reproduce faster than mortality. Rodda et al. (1999b) demonstrated that prey availability is a key parameter limiting brown treesnake success in the native range and in the Mariana Islands, and prey availability is likely to be a factor in ability to colonize North America. Prey availability is likely to be loosely and positively correlated with both temperature and rainfall, but the brown treesnake can transcend excessively cold or dry seasons by hibernation and aestivation, respectively. Hibernation and aestivation are exhibited by brown treesnakes in Australia (Ehmann 1992, Ehmann 1993, Greer 1997), but the durations are not known. Unless the unfortunate colonization of the mainland US occurs, we can not know whether the temperature limits that apply to the brown treesnake in its native range will apply equally in North America. Proceeding on the basis that they may, we delineated the monthly mean temperature and rainfall limits to determine which areas of the mainland US are at greatest risk of brown treesnake colonization.

## METHODS

We used the range map in Rodda et al. (1999a) to distinguish occupied from unoccupied localities. We pooled localities within each 1 degree latitude/longitude (lat/long) cell. These localities were matched to the closest choice from among the 85,000 stations reported in the World Climate (2007) data set. When elevation data were available (i.e., most cells) we made sure that weather station elevations matched snake localities. The World Climate stations were grouped into lat/long cells of 1 degree. Although the World Climate dataset was large, mean monthly temperature figures were unavailable for many of the cells in the brown treesnake's native range. When possible, we used single stations that reported mean monthly values for both rainfall and temperature. In a few cases, we combined data from two stations in the same cell to obtain both climate data types. In the vast majority of cases for which both climate data types were available, we were able to match snake presence records with weather stations located in the same 1-degree lat/long cell, but for a few important localities we could find weather records only for an adjacent cell (only weather stations with equivalent elevations were considered). World Climate has no stations appreciably above sea level for New Guinea, yet montane locations are occupied by the brown treesnake (Rodda et al.1999a). We were able to obtain two useful upper montane New Guinea climate records from Hyndman and Menzies (1990).

To analyze rainfall on a logarithmic scale and include weather stations that had not recorded rainfall during particular months, we coded zero rainfall means as 0.01 mm/mo. We were able to match one Guam and 38 native range localities with appropriate climate data: Australia (21), Indonesia (7), Papua New Guinea (8), Guam (1) and the Solomon Islands (2).

We plotted each of the 39 climate records as 12-sided polygons, each vertex representing the mean conditions for one month of the year. We anticipated that the aggregate climate space occupied by the 39 polygons would be reasonably well defined in relation to maximum heat and rainfall, but would have irregular excursions into climate space of extreme cold and aridity, representing months of hibernation and aestivation, respectively.

By progressively flagging the 1-3 months of greatest aridity against the graphical background of the 39 climate polygons, we observed that only the first- and second-most arid months were confined to sparsely-occupied climate space. From this, we inferred that the brown treesnake generally avoids extreme aridity, but is probably capable of up to 2 months of aestivation. We conducted a similar analysis for hibernation periods of 2-5 months, and concluded that brown treesnakes can probably hibernate for 4 months.

We fit the closest convex polygon that included all points believed to represent climatic conditions experienced by active brown treesnakes, and checked this climate envelope hypothesis against field observations reported in the literature.

Finally, we applied our hypothesized climate envelope implied by the 39 climate polygons from native range weather station data to climates recorded in the mainland US. We obtained average monthly precipitation (cm) and average monthly temperature (°C) data from the on-line Daymet database for the US (http://www.daymet.org; Thorton et al. 1997). Thorton et al. (1997) used daily observations from over 6,000 stations across the US collected from 1980 to 1997 to create the surfaces at a 1 $km^2$ resolution. The equations defining the climate space of the convex polygon

occupied by the 39 climate polygons were implemented using Visual Basic for Applications with ESRI's ArcGIS 9.0 ArcObjects to produce the US map of habitat suitability for the brown treesnake.

## RESULTS

The 12-sided climate polygons for the 39 sites occupied by brown treesnakes exhibited a fairly definite upper temperature limit corresponding to a mean monthly temperature of 32.5 C (Figure 1). Similarly, the upper rainfall limit was decisive; we obtained no evidence of brown treesnake occupancy of sites with mean monthly precipitation of > 1000 mm (Figure 1). The two high elevation sites from Papua New Guinea exhibited cool, wet conditions that did not overlap with the conditions reported from climatically adjacent sites (all of which were at or near sea level).

In contrast to the relatively compact climate polygons on the warm and wet sides of the brown treesnake climate space, the arid and cold sides exhibited short excursions into very dry and fairly cold conditions. If one excludes the arid excursions from the climate space that the brown treesnake can occupy and remain active, the range of monthly rainfall ranges from 2-1,000 mm.

Our climate envelope hypothesis outlined in Figure 1 indicates an active monthly mean temperature range 10-32.5 °C. This range of conditions is much broader than that experienced on Guam (Figure 1), which itself is a substantially wider range of conditions than is present on Manus, the island believed to be the source of the Guam population (Rodda et al. 1992). Manus has equable rainfall (monthly means 218-346 mm) and practically invariant temperature (monthly means 27.0-27.4 °C).

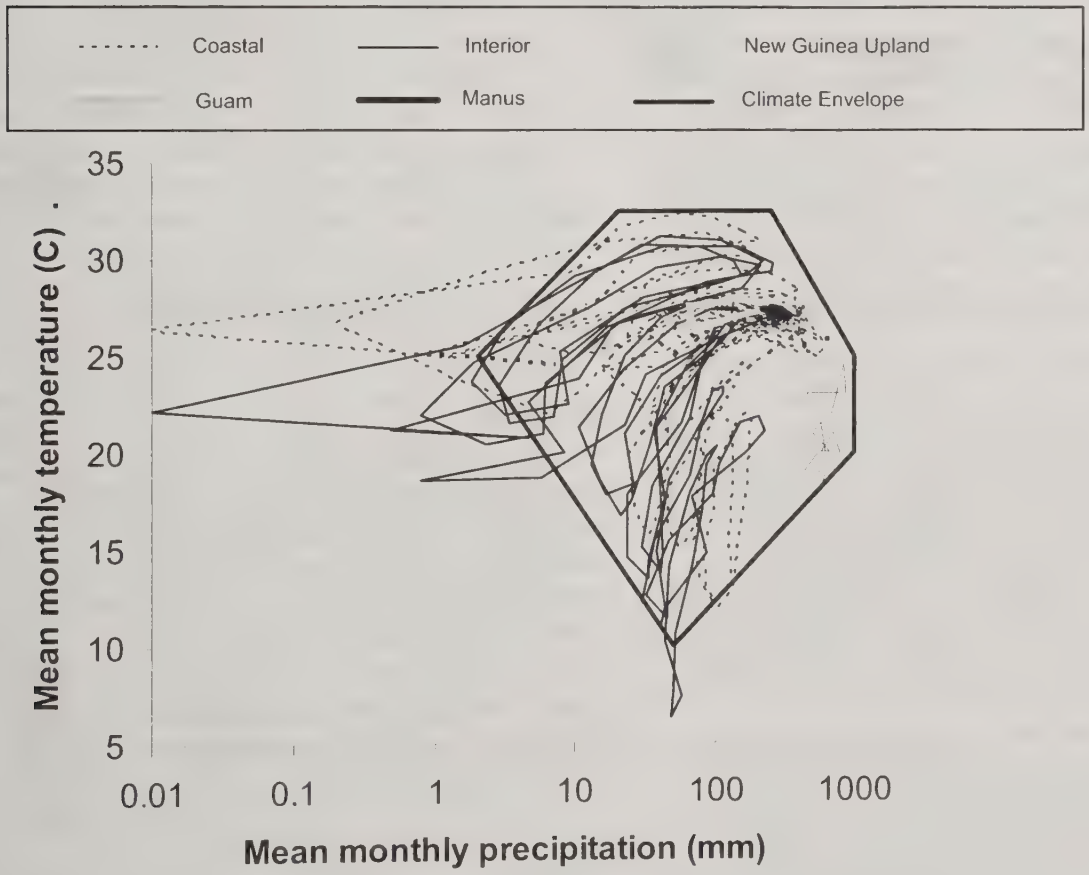



**Figure 1.** Climate envelope for the native range of the brown treesnake as delimited by mean monthly precipitation and temperature. All of the 12-sided polygons represent native range sites except Guam, where the snake is introduced but very successful. Manus is believed to be the origin of the population on Guam.

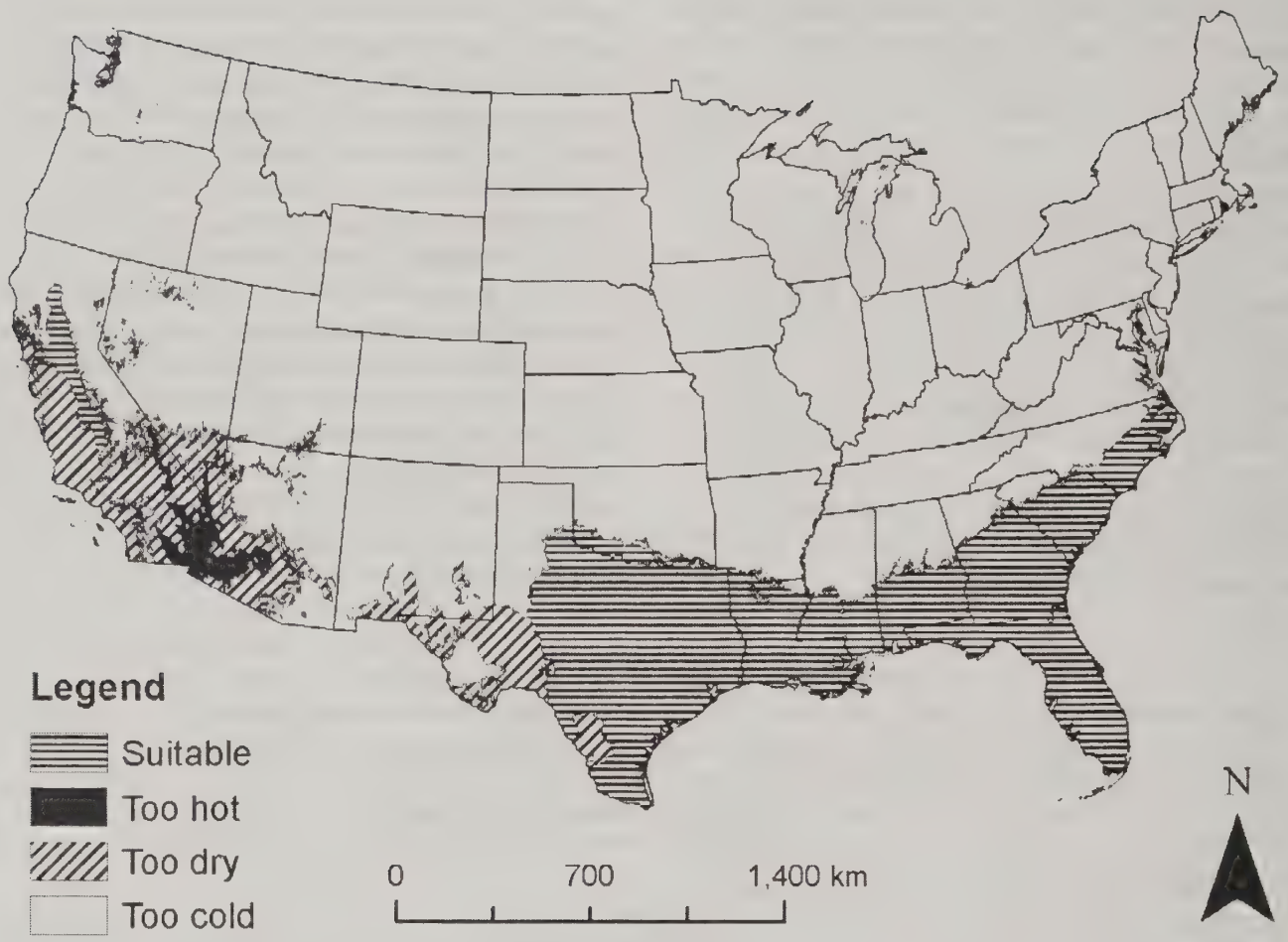



**Figure 2.** Potential brown treesnake range in the continental US, as delimited by the climate envelope shown in Figure 1.

Based on climate match, much of the continental US was deemed too cold for brown treesnakes (Figure 2). Nonetheless, a broad swath of the southeastern US coastal plain has a suitable climate, including most of (west to east) Texas, Louisiana, Florida, Georgia, and South Carolina, and substantial parts of (west to east) California (Central Valley), Mississippi, Alabama, and North Carolina. Desert areas of the Southwest were generally too dry (Mojave, Sonoran, and Chihuahuan deserts), and the lower Colorado River basin was seasonally hotter than the prescribed 32.5 °C. Nonetheless, scattered throughout and surrounding these zones of excess heat and aridity were slightly higher elevation sites that were acceptably mesic.

## DISCUSSION

Although weather data from the brown treesnake's native range exist for most coastal localities, upland weather records were conspicuously underrepresented. For example, if we had not obtained the two upland Papua New Guinea records, we would have missed a substantial portion of the occupied climate space. The disjunct location of those upper montane records on the climate space diagram (Figure 1) indicates that lower montane weather records are missing, despite the ample field evidence that brown treesnakes are not only present, but exceptionally abundant in lower montane sites (Parker 1982, O'Shea 1996, Rodda et al. 1999b). We note that even the wetter coastal localities were undocumented in the World Climate data set (e.g., the southern or "weather" coast of Guadalcanal Island had no stations and no European settlements), perhaps reflecting a preference of

European settlers for drier colony sites within Melanesia. If such sites were climatically documented, we expect that the upper acceptable rainfall limit for brown treesnakes would be expanded substantially beyond the 1000 mm/month we recorded. Such an expansion would not affect the identification of suitable areas of the mainland US, but would increase the range of sites deemed suitable in Asia and South America.

The range of sites deemed suitable in North America uniformly includes all sea ports from the Mexican border of Texas to Norfolk, Virginia (Figure 2), and most destinations within the Southeastern coastal plain. The situation on the West Coast is complicated by the interdigitation of acceptable and unacceptable sites in southern California, though the Central Valley possesses a uniformly suitable climate. Thus shipments into or via San Francisco would be at higher risk than those destined for Los Angeles or Phoenix. Shipments destined for montane sites in southern California or Arizona might be at risk if the elevation is moderate (Figure 2). Furthermore, assignment of "too dry" to areas in the Southwest is sensitive to uncertainty about the permissible duration of aestivation in brown treesnakes. Direct evidence of the duration of aestivation in Australia would be useful for refining area boundaries in the American Southwest. Rocky Mountain, Northwest, North Central, and Northeastern states appear to be uniformly too cold for brown treesnakes. According to 2000 census figures, about 80 million people live in US counties where all or part of the county was classified as having a suitable climate.

Although the northern states are outside of the climate space inferred from the brown treesnake native range, the full scope of the snake's climate tolerances may not be reflected in the native range. The high latitude limit of the brown treesnake's native range is on a narrow front around Sydney, Australia, a relatively arid location wedged between saltwater and mountains. There are no cool high-rainfall sites at the high-latitude periphery of the brown treesnake's native range; thus caution should be applied in inferring the snake's cold-mesic tolerances.

The success of any invader is set by the population's ability to reproduce fast enough to offset mortality. Climate influences both reproduction and mortality, but the linkage is often indirect. None of the extant climate-matching programs goes much beyond basic climate statistics, and none measures recruitment or mortality rates. Although climate is an important correlated factor in invasion success, brown treesnakes have a variety of behavioral mechanisms (aestivation, hibernation, restricting activity to suitable microenvironments) for buffering themselves from physiologically-stressful environments. However, if insufficient prey resources are available, no amount of physiological tolerance will provide the energy required for reproduction. We do not adequately understand the operation of the complex interaction between a population and its environment, in particular that of brown treesnakes, nor are we able to predict how the interaction might change in a new locality or for a new population.

New colonizations often pass through a genetic bottleneck that alters the population's genetic makeup and potentially the population's realizable niche. The lineage of brown treesnake that occurs on Guam may have passed through genetic bottlenecks during the colonization of Guam and during the earlier colonization of Manus, which has a very restricted climate space (Figure 1). Laboratory tests of survival (Christy et al. 2007) indicate that Guam brown treesnakes possess a short-term temperature tolerance that is substantially wider than the range of monthly means shown in Figure 1 for all occupied sites; thus despite any putative genetic bottlenecks Guam snakes can survive the full range of conditions. Nonetheless, short-term survival may not be a good indicator of the environmental conditions required for invasion success.

Thus, there are a multitude of mechanisms that might allow brown treesnakes to occupy more or less of North America than suggested by this climate matching model. Based on climatic conditions, a substantial portion of the southern US is at risk of brown treesnake colonization. We did not investigate climate matching of Mexico and Neotropical sites, but our climate match indicates that the brown treesnake should be able to colonize areas immediately south of the Rio Grande river.

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## LITERATURE CITED


ARAUJO, M. B. AND M. NEW. 2007. Ensemble forecasting of species distributions. Trends in Ecology and Evolution 22:42-47.

BOMFORD, M., F. KRAUS, M. BRAYSHER, L. WALTER AND L. BROWN. 2005. Risk assessment model for the import and keeping of exotic reptiles and amphibians. Bureau of Rural Sciences, Canberra, ACT, Australia.

BROENNIMANN, O., U. A. TREIER, H. MÜLLER-SCHÄRER, W. THUILLER, A. T. PETERSON, AND A. GUISAN. 2007. Evidence of climatic niche shift during biological invasion. Ecology Letters 10:701-709.

CHRISTY, M., R. BISCHOF, G. RODDA, AND J. SAVIDGE. 2007. Can temperature be used as a tool for limiting brown treesnake invasion via transportation pathways? Pages 246-256 *in* G. W. Witmer, W. C. Pitt, and K. A. Fagerstone, editors. Proceedings of the managing vertebrate invasive species. August 2007. Fort Collins, Colorado, USA.

EHMANN, H. 1992. Encyclopedia of Australian animals: reptiles. Angus and Robertson, Pymble, New South Wales, Australia.

EHMANN, H. 1993. Family colubridae. Pages 290-294 *in* C. J. Glasby, G. J. B. Ross, and P. L. Beeseley, editors. Fauna of Australia. Vol 2A Amphibia and Reptilia. Australian Govt. Publ. Serv., Canberra.

ELITH, J., C. H. GRAHAM, R. P. ANDERSON, M. DUDÍK, S. FERRIER, A. GUISAN, R. J. HIJMANS, F. HUETTMANN, J. R. LEATHWICK, A. LEHMANN, J. LI, L. G. LOHMANN, B. A. LOISELLE, G. MANION, C. MORITZ, M. NAKAMURA, Y. NAKAZAWA, J. M. OVERTON, A. T. PETERSON, S. J. PHILLIPS, K. RICHARDSON, R. SCACHETTI-PEREIRA, R. E. SCHAPIRE, J. SOBERÓN, S. WILLIAMS, M. S. WISZ, AND N. E. ZIMMERMANN. 2006. Novel methods improve prediction of species' distributions from occurrence data. Ecography 29:129-151.

FRITTS, T. H. AND G. H. RODDA. 1998. The role of introduced species in the degradation of island ecosystems: a case history of Guam. Annual Review of Ecology and Systematics 29:113-140.

GREER, A. E. 1997. The biology and evolution of Australian snakes. Surrey Beatty & Sons, Chipping Norton, New South Wales, Australia.

GUISAN, A., A. LEHMANN, S. FERRIER, M. AUSTIN, J. MC. C. OVERTON, R. ASPINALL, AND T. HASTIE. 2006. Making better biogeographical predictions of species' distributions. Journal of Applied Ecology 43:386-392.

HYNDMAN, D. C. AND J. I. MENZIES. 1990. Rain forests of the Ok Tedi headwaters, New Guinea: an ecological analysis. Journal of Biogeography 17:241-273.

NIX, H. 1986. A biogeographic analysis of Australian elapid snakes. Pages 4-15 *in* R. Longmore, editor. Snakes; atlas of elapid snakes of Australia. Australian Bureau of Flora and Fauna , Australian Flora and Fauna Ser. No. 7, Canberra, Australia.

O'CONNOR, R. J. 2002. The conceptual basis of species distribution modeling: time for a paradigm shift? Pages 25-33 *in* J. M. Scott, P. J. Heglund, M. L. Morrison, J. B. Haufler, M. G. Raphael, W. A. Wall, and F. B. Samson, editors. Predicting species occurrence; issues of accuracy and scale. Island Press, Washington D.C., USA.

O'SHEA, M. T. 1996. A guide to the snakes of Papua New Guinea. Independent Pub., Port Moresby, Papua New Guinea.

PARKER, F. 1982. The snakes of the Western province. Wildlife of Papua New Guinea 82:32-33.

PHELOUNG, P. C. 1996. CLIMATE: a system to predict the distribution of an organism based on climate preferences. Agriculture Western Australia, Perth, Australia.

RODDA, G. H., T. H. FRITTS, AND P. J. CONRY. 1992. Origin and population growth of the brown tree snake, *Boiga irregularis*, on Guam. Pacific Science 46:46-57.

RODDA, G. H., T. H. FRITTS, M. J. MCCOID, AND E. W. CAMPBELL, III. 1999a. An overview of the biology of the brown treesnake, *Boiga irregularis*, a costly introduced pest on Pacific Islands. Pages 44-80 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the Habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

RODDA, G. H., T. H. FRITTS, M. J. MCCOID, AND E. W. CAMPBELL, III. 1999b. Population trends and limiting factors in *Boiga irregularis*. Pages 236-253 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

SAVIDGE, J. A. 1987. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660-668.

SCOTT, J. M., P. J. HEGLUND, M. L. MORRISON, J. B. HAUFLER, M. G. RAPHAEL, W. A. WALL AND F. B. SAMSON. 2002. Predicting species occurrence; issues of accuracy and scale. Island Press, Washington D.C., USA.

STOCKWELL, D. R. B. 1999. Genetic algorithms II. Pages 123-144 *in* A. H. Fielding, editor. Machine learning methods for ecological applications. Kluwer Academic Publishing, Boston, Massachusetts, USA.

STOCKWELL, D. R. B. AND I. R. NOBLE. 1992. Induction of sets of rules from animal distribution data: a robust and informative method of analysis. Mathematics and Computers in Simulation 33:385-390.

STOCKWELL, D. R. B. AND D. PETERS. 1999. The GARP modeling system: problems and solutions to automated spatial prediction. International Journal of Geographic Information Science 13:143-158.

STOCKWELL, D. R. B., SHANER, S., TATE, D. J., BORK, K., AND S. DOUGHERTY. 2005. Integrating transportation network data into predictive models for invasive species. Report to U.S. Department of Transportation, 15 Nov 2005, Washington D.C., USA.

SUTHERST, R. W., R. B. FLOYD, AND G. F. MAYWALD. 1996. The potential geographical distribution of the cane toad, *Bufo marinus* in Australia. Conservation Biology 10:294-299.

THORTON, P.E., S. W. RUNNING, AND M. A. WHITE. 1997. Generating surfaces of daily meteorology variables over large regions of complex terrain. Journal of Hydrology 190:214-251.

WORLD CLIMATE. 2007. www.worldclimate.com accessed on numerous occasions in Jan/Feb. 2007.

# THE ROLE OF EARLY DETECTION AND RAPID RESPONSE IN THWARTING AMPHIBIAN AND REPTILE INTRODUCTIONS IN FLORIDA

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***Abstract:*** Prevention is the best policy for dealing with introduced species. However, biologists often spend an inordinate amount of time studying their spread and impacts rather than focusing on what should be done to thwart their establishment in the first place. Amphibian and reptile introductions are reaching epidemic proportions in Florida, largely due to irresponsible behavior by pet owners and the pet industry, but also due to ineffective preventive policies and actions. Prevention of additional amphibian and reptile introductions in Florida will require a comprehensive approach involving legal restrictions of certain problematic species, a massive public education effort, and a well-funded and staffed Early Detection and Rapid Response (EDRR) program. EDRR is not a novel concept, but it needs to be newly applied to amphibian and reptile introductions where pathways are firmly established and propagule pressure is intense. An effective EDRR program in Florida will require (1) significant funding and political will, (2) a comprehensive stakeholder education and public outreach program, (3) a vast network of expert early detectors, (4) a team of talented rapid responders, and (5) rigorous post-project assessment. Knowledge gained from such a program in Florida could easily be extended to other taxonomic groups and locations.

***Key Words:*** amphibian, early detection, eradication, Florida, introduced, invasive species, prevention, rapid response, reptile.

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## INTRODUCTION

Introduced species are a form of biological pollution; however, they represent a problem for which dilution is not a solution. Unlike most chemical pollutants, introduced species often become established long before their presence is known, and they also may expand exponentially, resulting in a problem that worsens over time. Once they are firmly established, the range of options for dealing with non-indigenous species changes considerably (Simberloff et al. 2005). As a result, prevention is widely considered the most effective, efficient, and economically viable policy for dealing with introduced species (Pimentel 2002).

Effective prevention requires a multifaceted approach, including (1) legal restrictions on the import, breeding, ownership, and sale of problematic species, (2) quarantine of illegally imported problematic species, (3) effective public education about introduced species problems, (4) early detection of incipient populations, (5) swift, decisive action to eradicate incipient populations, and (6) risk assessments for potential future invaders. Specific protocols for dealing with many introduced species in the early stages of their establishment are surprisingly underdeveloped (Myers et al. 2000), especially when multiple populations and import pathways are involved (Lockwood et al. 2007). This is particularly true for amphibian and reptile introductions, which have reached epidemic proportions in Florida due to irresponsible behavior by the pet industry and their customers, delays in reporting and lack of follow-up by scientists in the field, and ineffective governmental policies and actions for preventing new introductions. In this paper, I will (1) summarize the extent of the problem, (2) identify some of the roadblocks that prevent us from finding solutions, and (3) provide a framework for an Early Detection and Rapid Response (EDRR) program that could help thwart the establishment of additional non-native amphibians and reptiles in Florida and beyond.

## WELCOME TO FLORIDA!

Florida's sub-tropical climate makes it an ideal place for the establishment of non-native amphibians and reptiles from all over the world. The Sunshine State is also a haven for the amphibian and reptile pet industry, private herpetoculturists, and pet owners, many of which

house their pets in outside cages and open enclosures. While hurricanes and careless import and husbandry practices result in some accidental releases, well-intentioned pet owners often release unwanted pets into suitable habitats, and unethical breeders may release multiple individuals in order to establish new populations as a future source of animals. As a result, Florida is one of the most invaded places on Earth.

Along with over 2,000 introduced pathogens, plants, invertebrates, fish, birds, and mammals (Simberloff et al. 1997), at least three amphibians and over 40 reptiles are now established in Florida (Meshaka et al. 2004). Notorious species include the cane toad (*Bufo marinus*), Cuban treefrog (*Osteopilus septentrionalis*), Cuban brown anole (*Anolis sagrei*), green iguana (*Iguana iguana*), Mexican spiny-tailed iguana (*Ctenosaura similis*), Nile monitor (*Varanus niloticus*), and Burmese python (*Python molurus*). In fact, there are now more non-native lizards established in south Florida than there are native lizards in the entire southeastern United States (US), and whole assemblages of non-native lizards (anoles, geckos, iguanids, and teiids) can be observed at some locations (e.g., most botanical gardens in Miami). With so many introduced amphibians and reptiles using mainly introduced plants as habitat and consuming mainly introduced plants, arthropods, and vertebrates, Florida is a grand example of invasional meltdown (Simberloff and Von Holle 1999). How did we get to this point?

## NERO IS FIDDLING WHILE FLORIDA IS BURNING

The first introduced amphibians and reptiles (IAR) arrived in Florida over a century ago, and early publications on the herpetofauna of Florida contained information about 7 established species (Carr 1940, Carr and Goin 1955, Duellman and Schwartz 1958). King and Krakauer (1966) wrote the first comprehensive summary of the problem, identified the pathways and potential impacts of 13 species (10 lizards and 3 anurans), issued a stern warning that the native fauna would suffer if introductions continued, and even called for preventative measures. Twelve years later, Smith and Kohler (1978) discussed the IAR problem throughout the US and for 16 species in Florida. By the early 1980s, Floridians had anecdotal evidence their native green anole (*Anolis carolinensis*) was being replaced by the Cuban brown anole in urban and residential areas (Campbell 2000). Wilson and Porras (1983) included detailed accounts of newly established species and lengthy discussions of potential impacts, but underplayed the impacts of IARs relative to habitat alteration. Moreover, they argued that eradication was not morally defensible, and even suggested that Floridians should accept them as part of a new urban herpetofauna (their emphasis), utilize them for biological materials, establish a quota system for their harvest, document their arrival, and conduct more research on their impacts and competitive effects, rather than develop prevention, eradication, or management plans.

By the 1990s, hundreds of new distribution records and natural history notes documenting new IARs and numerous studies revealing impacts of established species were published in Herpetological Review and other peer-reviewed journals. Meanwhile, the popularity of amphibians and reptiles as pets increased rapidly due to advances in medicine and husbandry techniques and heavy promotion by the pet industry and private breeders, and the number of introductions accelerated. Dalrymple (1994) revealed the extent of the growing IAR problem on public lands in south Florida, and Butterfield et al. (1997) summarized the overall problem in detail, but like Wilson and Porras (1983), they touted IARs as merely a symptom of habitat alteration, and even predicted that IARs would not invade or negatively affect native species or natural habitats in Florida.

Unfortunately, the erroneous perception that IARs in Florida were either benign or would only colonize and affect human-altered landscapes was repeated so often that it became common knowledge. Certainly, one would expect invasions to be more common in areas populated by humans, but it was probably reckless to assume, without supporting data, that no IARs of any consequence would expand beyond human habitation or affect native species. There was even a general reluctance to view the establishment of small, abundant species (e.g., *Anolis*) as potentially problematic (Butterfield et al. 1997), despite the extraordinary densities and biomass of the brown anole (Campbell and Echternacht 2003), the near complete replacement of the green anole in urban areas despite its long association with even the most urban landscapes (Campbell 2000), and its dramatic effects on Bahamian food webs (Schoener and Spiller 1999). Decades of peer-reviewed and popular publications about IARs in Florida merely instigated more basic studies of their distribution,

abundance, impacts, and evolution, rather than steer biologists towards applied questions such as prevention, eradication, or management.

To-date, no established IAR species has ever been eradicated from Florida. In fact, only 6 IARs have been the subject of organized eradication or management efforts of any kind. Burmese pythons have spread widely in the Everglades, but biological studies and trapping efforts have only just begun (S. Snow, personal communication). Argentine tegu lizards (*Tupinambis rufipunctatus*) are established near Tampa, but their impacts are unknown, and preliminary trapping efforts have been unsuccessful to-date (B. Kaiser, personal communication). Green iguanas are abundant throughout south Florida, and localized trapping and management efforts have not measurably reduced iguana populations (Krysko et al. 2007). Mexican spiny-tailed iguanas are a serious problem on the beaches of Gasparilla Island (Krysko et al. 2003), and the County government instigated a massive trapping effort, bounties, and even an "iguana tax", but the effort is on-going and thousands of lizards remain. Nile monitor lizards are widespread and problematic in Cape Coral (Enge et al. 2004) and results from a preliminary trapping study and eradication effort are encouraging (Campbell 2005), but thousands of lizards remain, the population is rapidly expanding into natural areas, and new populations have been discovered elsewhere. Finally, results from two assessments of the utility of PVC pipe refugia in reducing Cuban treefrog populations (Rice et al. 2003, T. Campbell, unpublished data) are encouraging, but in terms of the operational viability of managing this widespread, abundant species using PVC pipe refugia in natural areas, the jury is still out. Ultimately, all six of these species were firmly established when management efforts began, and none of these efforts have received the level of funding, agency attention, or political support that would ensure success.

In summary, agency, industry, and academic biologists have known about the worsening IAR problem in Florida for over 6 decades, but the warnings issued every decade since 1940 were muted and have gone largely unheeded. Until the recent arrival of a few large predatory species, IARs were simply not seen as a problem worth our attention beyond basic biological and population studies. Indeed, there is a limit beyond which further biological studies of an introduced species do not assist in its eradication (Simberloff 2003). Given decades of complacency and misdirected research, it is no surprise that a recent comprehensive summary of the IAR problem in Florida (Meshaka et al. 2004) required a bound volume and painted a very dreary picture. As invasional meltdown proceeds in Florida, biologists are beginning to understand that the Emperor has no clothes: that IARs of all taxa and sizes potentially represent significant threats to the native species that nature preserves were designed to protect, and that future invasions should be thwarted whenever possible. But how might this be accomplished?

## EARLY DETECTION AND RAPID RESPONSE: A PARTIAL SOLUTION

Despite our best intentions, prevention efforts will never thwart all introductions. Aggressive, coordinated intervention is often necessary to prevent incipient populations from spreading beyond the point where eradication is economically and logistically feasible (National Invasive Species Council 2001, 2003). This is where Early Detection and Rapid Response (EDRR) comes into play. Success of an intervention depends on the speed and veracity of the response, which is in turn dependent on coordination of stakeholders. Effective EDRR requires careful coordination of two separate but interrelated phases. Early Detection (ED) reveals the extent of the problem and assesses the potential for a rapid and successful eradication. Early detection and reporting is critical given the speed with which many IAR populations expand numerically and spatially. Rapid Response (RR) is a carefully planned, decisive action designed to eradicate the incipient population. Together, these efforts ultimately serve to prevent an incipient population from expanding beyond the point where it can be eradicated quickly and efficiently (National Invasive Species Council 2001, 2003). However, EDRR is not a long-term management or control strategy. If an EDRR effort is unsuccessful, the response team should provide information to another group of managers for long-term management or control and quickly move on to the next incipient species.

There is no need to reinvent the wheel when designing an EDRR program for IARs in Florida. A plethora of EDRR programs, networks, and initiatives have been mandated or established for many taxa at the international, federal, regional, state, and local levels all over the world. These templates could be easily altered to fit the special problems of IARs and unique operational issues in

Florida. I review a sample of government reports, peer-reviewed publications, and web sites that describe EDRR efforts in the US, critique their efficacy, and make suggestions for improvements. A much more detailed, but potentially dated, review of these programs is provided in Worrall (2002), and important documents are available at www.invasivespecies.gov.

Starting primarily with the Lacey Act in 1900, the US has a long history of introduced species prevention, eradication, and management. Over 20 federal agencies have at least some responsibilities for introduced species management (General Accounting Office 2001). However, the need for a national system for detecting, responding to, and monitoring incipient populations has only recently been realized. Executive Order 13112, issued by President Clinton in 1999, requires federal agencies to detect and respond to incipient populations in a cost-effective manner. The National Invasive Species Council (NISC) generated a National Invasive Species Management Plan with specific recommendations for EDRR in the US (NISC 2001). Still, the obstacles hindering rapid response efforts at the federal level are extensive (General Accounting Office 2001). Rapid response to introduced agricultural pests is often intense and effective, but the response to invasive species in natural areas has been minimal (General Accounting Office 2001). In FY 2000, nearly $150 million was spent on RR efforts in the US. Over 90 percent of these funds was spent on species that affect agricultural and silvicultural species and pests of native trees, and for a number of aquatic nuisance species, but no amphibians, and only one reptile, the brown treesnake (*Boiga irregularis,* BTS), appears on the list. Although agricultural and food pests have been a high priority for good reason, a more balanced approach is now clearly warranted.

The Federal Interagency Committee for the Management of Noxious and Exotic Weeds (FICMNEW) published a conceptual design for a national EDRR system for introduced plants in the US (NISC 2003). This document was the result of 5 years of planning with extensive agency, industry, and public input. Although the proposed EDRR System is specific to plants, it has vast applicability to IARs in Florida. They elaborate on 5 specific EDRR System elements: detection and reporting, identification and vouchering, rapid assessment, planning, and rapid response (including post-response assessment). They also provide a framework for coordination, support, oversight, information management, stakeholder involvement, and outreach, and identify the resources that are needed to establish their proposed system.

The Aquatic Nuisance Species (ANS) Task Force is an interagency committee charged with developing and implementing a program to prevent the introduction and dispersal of non-native aquatic species in waters of the United States (Parker and Keeney 2004). The ANS Program document and ANS Strategic Plan include a number of prevention strategies and established 5 committees with species-specific Working Groups. The Communication, Education, and Outreach Committee approaches this problem from the perspective that aquatic introduced species are not readily observed, thus, their negative effects are not realized until their populations are firmly established. This is also true for IARs in Florida. Also, the ANS Task Force Control Committee is one of the agencies charged with controlling the BTS in Guam and other areas where it has been introduced, and preventing its introduction to the North American mainland (Parker and Keeney 2004). Thus, the ANS Task Force already has experience dealing with at least one terrestrial reptile from an EDRR perspective.

The ANS Task Force is also charged with educating the public about the problems of introduced species and empowering people to actively prevent their establishment; both essential elements of a successful EDRR program. The ANS Task Force's Stop Aquatic Hitchhikers!™ campaign has a vast infrastructure of partner organizations and a consistent educational message that is broadcast in many different formats (Parker and Keeney 2004). The ANS Task Force's Habitatitude™ campaign (www.habitatitude.net) recruits the aquarium and aquaculture industry, governmental agencies, and academia to proactively educate aquarists and consumers of ornamental aquatic plants about the risks and negative effects of aquatic introduced species, and provide alternatives for releasing unwanted species into the wild (Parker and Keeney 2004). The Habitatitude™ campaign has more recently incorporated IARs into their message, and has been proactive in working with agencies, industry, academia, reptile wholesalers and breeders, and private herpetoculturists in Florida. The educational and outreach components of these programs mainly address prevention of releases, but will be essential for the establishment of an effective EDRR program for IARs in Florida.

In Florida, many iterations of EDRR have long been practiced by exotic plant and agricultural pest managers, but this concept was rarely mentioned as an operational strategy in a comprehensive volume published only a decade ago (Simberloff et al. 1997). In fact, while 9 state agencies are responsible for prevention and management of invasive non-native species, only 3 have regulatory authority (NISC 2003), there is no interagency coordination mechanism for prevention or management (NISC 2003), and there is no state agency charged with the operational aspects of non-indigenous wildlife management (Ferriter et al. 2006). However, recent developments are encouraging.

Florida's Invasive Species Working Group (ISWG), formed in 2001 by personnel from a number of agencies, included rapid response as an action item in their Statewide Strategic Plan for Florida (ISWG 2003). They committed to recommending RR procedures and improved coordination with federal and local agencies. In 2004, the Florida Invasive Animal Task Team (FIATT) was formed by the South Florida Ecosystem Restoration Task Force to deal with introduced species issues affecting the Comprehensive Everglades Restoration Project (CERP). They generated species list for each taxon from which priorities can be set by triage. On a positive note, 2 mammal invasions may have been recently thwarted by effective EDRR programs (R. Engeman and B. Constantine, personal communication). These kinds of successes will be extremely valuable for generating funding for EDRR efforts in Florida (L. Williams, personal communication).

Despite the vast number of vertebrate EDRR programs in place worldwide, very few have been initiated for IARs. Much can be learned from the rich literature on the well-funded, multi-agency efforts to eradicate the BTS from Guam and thwart its spread to other South Pacific islands, Hawaii, and beyond (Rodda et al. 1999, Parker and Keeney 2004). Efforts to control the Puerto Rican coqui frog (*Eleutherodactylus coqui*) in Hawaii resemble EDRR on a local scale, but this species spread beyond EDRR due to a lack of funding, governmental delays, and disbelief in the problem (Kraus and Campbell 2002). In Florida, despite decades of warnings and the rapid accumulation of nearly 50 IARs, only a handful of management programs have been initiated, and all were clearly well beyond EDRR, but much can be learned from those efforts.

Stakeholder involvement and coordination efforts involving EDRR have gained momentum in Florida in recent years. Prevention and management was featured during the "Introduced Amphibians and Reptiles: From Case Studies to Solutions" Symposium at the 2005 Joint Meeting of Ichthyologists and Herpetologists in Tampa, Florida. Specific strategies were developed by the Early Detection and Rapid Response Working Group at the 2005 Invasive Snake/Reptile Management and Response Workshop in West Palm Beach, Florida. Pet industry and Habitatitude! representatives, agency personnel, legislators, managers, academics, herpetoculturists, and other stakeholders gathered at the 2006 All-Florida Herpetology Conference in Gainesville, Florida, to discuss EDRR and pending legislation restricting ownership of large reptiles. Soon after, I vetted these ideas at a Central Florida Herpetological Society meeting, and received mostly positive feedback. Recent meetings of the ISWG and FIATT featured specific discussions about potential EDRR program funding, staffing, and infrastructure. Many of the key participants of the above meetings and workshops continue to place EDRR high on their priority list and are working hard to generate interest and obtain funding.

## AN EDRR PROGRAM PROPOSAL FOR INTRODUCED HERPS IN FLORIDA

Detection, assessment, and response are generally viewed as the minimum components of any successful EDRR program (NISC 2003). I propose that an effective EDRR Program for IARs in Florida will require (1) significant political will, long-term funding, and governmental agency infrastructure, (2) a comprehensive stakeholder education and public outreach program, (3) a network of informed early detectors, (4) a team of talented rapid responders, and (5) rigorous post-project assessment.

### Political Will, Long-Term Funding, and Infrastructure

Stakeholders should be responsible for generating the motivation and political will to implement such a program. This will require frequent interactions with legislators and agency personnel and a massive public education campaign. Because the EDRR task at hand is immense and potentially never-ending, it will require significant funding, in perpetuity.

Government agencies generally have experience with the operational aspects of pest management and have the funding and required tools. Thus, I believe the funding and infrastructure for an EDRR program should be established by a collaborative group of federal and state agencies, such as the ANS Task Force, ISWG, Florida Fish and Wildlife Conservation Commission (FWCC), Florida Department of Environmental Protection (FDEP), and FIATT.

Once the political will is generated, it will be up to the agencies to determine how to obtain funds, set up the infrastructure, and partition the workload. Funds should be used to establish an infrastructure that includes at least (1) a dedicated 1-800 phone number, (2) an informative, interactive web site, (3) a centralized information repository, (4) a network of early detectors and taxonomic experts, and (5) a team of rapid responders. However, this infrastructure will only be effective if it is linked directly to an aggressive education and outreach campaign.

**Stakeholder Education and Outreach**

An effective education and outreach campaign is critical to the success of any EDRR program. Examples of proactive strategies with eye-catching, positive messages include the ANS Stop Aquatic Hitchhikers! and Habitatitude campaigns, the BTS campaign (N. Hawley and C. Martin, personal communication), the Burmese python campaign (S. Snow, personal communication) and Hawaiian Ecosystem At Risk (HEAR) project. This paper is not intended to discuss the creation and administration of introduced species education campaigns in general. Rather, with regards to EDRR, results from these programs suggest stakeholders must (1) understand the problem and have a reason to care, (2) know that an EDRR program exists and is worth using, (3) have at least a rudimentary knowledge of which species are native and which are introduced, and (4) know who to contact to report a sighting or capture of something they cannot identify.

The first line of defense against the establishment of new herpetiles will be information and images provided by informed observers and early detectors, including the public (e.g., hikers, birders, photographers), professional field biologists (e.g., agency, NGO, and industry biologists), and academic biologists. However, not everyone can be expected to correctly identify an amphibian or reptile they have not seen before. They may only believe it to be somehow different than anything they have ever seen. Most often, the observer will only be able to manage a vague description of the critter (e.g., "a big, dark, stripped lizard walking across my yard"). On balance, the proliferation of cell phone cameras may benefit our cause, but in my experience the images usually resemble the Sasquatch on the National Enquirer. It will be up to the experts to filter out the noise.

To accurately identify non-native species, stakeholders and experts must have access to information about native and introduced species. Meshaka et al. (2004) is the most recent comprehensive source for identification and known localities of existing IARs in Florida, but despite their Herculean effort, it was out of date by the time it was published. Rapid senescence will be a problem for any information source published on paper. Bio-profiles (K. Enge, personal communication) made available in electronic format on-line are a powerful way to disseminate information, and may even result in published manuscripts (Enge et al. 2004). It might be more helpful to establish a centralized repository of information about the native and introduced amphibians and reptiles in Florida (e.g., the current FWCC web site). Ideally, the repository would also include information about popular pets that might escape or be released. In any event, a team of herpetologists should be available to verify the accuracy of sightings by the public.

For EDRR to work, informed observers, early detectors, and the general public must either know or be able to easily determine how to use the infrastructure to report a sighting. First, a toll-free phone line should be established for reporting sightings, and this phone line should have a live operator recording information and routing calls or an automated message system that is checked regularly. Phone contact information should be routed to the appropriate entities which the public is most likely to contact, including federal agencies, state agencies, and all county-level animal control and law enforcement agencies.

More importantly, the public should be able to easily find out how to report a sighting using an Internet search engine. Search results should take the observer directly to a central web site established specifically for this effort. A talented web writer will be able to insert a series of "meta tags" that will help to bring the Amphibian and Reptile Rapid Response Group (ARRRG) site to the top of the list of search results. The site itself should be very user-friendly in order to minimize frustration by less Internet-savvy users. It should

not only provide key EDRR team member contact information (e-mail and phone numbers), it should enable observers to directly enter their sightings and submit images using an on-line observation form, and those submissions should be reviewed on a regular basis. This web site should be mounted on a state-level server (e.g., FWCC) to ensure stability.

The media, which includes newspaper, magazines, television, Internet, and radio, can be a very powerful tool for education and outreach when used properly. I utilized local newspapers and television with much success to inform the public about Nile monitor lizards in Cape Coral and generate sightings (Campbell 2005). The fact that the story reached many media outlets beyond Cape Coral was an unintended, but largely beneficial consequence. Of course, there are many down-sides to the active use of the media in science; however, the benefits will generally outweigh the costs. We should not expect the general public to be experts. Rather, we should empower them to learn which species are non-native and problematic, and give them the means to contact the experts that are trained to deal with those species.

**The Amphibian and Reptile Rapid Response Group (ARRRG)**

A network of early detectors, taxonomical experts, and rapid responders will be required to thwart the establishment of incipient populations of IARs in Florida and beyond. Although the ISWG already has their FIATT, that team is specific to the South Florida region, is administered by CERP, and deals with any invasive animal taxa. I propose the establishment of the ARRRG, a team that works specifically on IARs. With its pirate connotations, this acronym has a high probability of being remembered by stakeholders, an important aspect of any outreach program (N. Hawley, C. Martin, and D. Vice, personal communication). Although the ARRRG would initially be specific to Florida issues, Florida was left out of the team name because I envision this group will eventually work nationwide.

The ARRRG should be composed of a network of individuals ranging in taxonomic and conceptual expertise from expert laypersons (hobbyists) to professional biologists. Professionally, the ARRRG should be populated by relevant personnel from government agencies, non-governmental organizations, academia, and industry, as well as the public. University of Florida Institute for Food and Agricultural Sciences (UF-IFAS) Extension Offices are present in nearly every county in Florida, already deal with exotic plant issues, possess the infrastructure to coordinate and share information among stakeholders, and directly involve the public in many activities and courses, so could play an important role in the IAR-EDRR Program (S. Johnson, personal communication).

To increase efficiency, the ARRRG should be partitioned into overlapping functional groups including early detectors, taxonomic experts, and rapid responders. Early detectors are the informed observers at the front lines of the EDRR effort. They must have their eyes and ears continually trained on the field, must possess just enough background in natural history and taxonomy to quickly detect the arrival of a non-native species, must be able to photograph, capture, or otherwise document their observations, and must know exactly how to report their observations. Suitable spatial coverage is important for early detection, so informed observers must be present in higher densities at locations where pathways are established (e.g., near ports, wholesalers, and breeding operations), but should also be present in more remote areas. Early detectors might include agency, NGO, industry, and academic biologists, bird watchers, fishing guides, landscapers, fire fighters, or anyone that regularly works outside. Given the establishment of an effective education and outreach program, the general public may even serve as part of this network.

Once a novel species is detected and either photographed or captured, taxonomic experts from agencies, natural history museums, academia, or the pet industry should be recruited to verify the early detectors' identification before proceeding with a response. Ideally, these experts would have the knowledge or access to pertinent literature to generate a brief synopsis of the problem, including the potential for establishment, numerical expansion, spatial spread, and feasibility of eradication of the incipient population. The synopsis could range in style from a simple e-mail to a formal agency report, bio-profile, or draft manuscript for publication. Most importantly, this information should be immediately disseminated to a team of biologists that stand ready to quickly assess the problem and formulate a rapid response plan.

The rapid response team should be responsible for operational aspects, including finding the population, assessing the magnitude of the problem, and attempting to capture, trap, or otherwise quickly eradicate the incipient population. The

time it takes to complete these tasks will depend on many biological characteristics of the pest species, including the population size, fecundity, behaviors and activity periods, movements, and ease of capture and handling. Hundreds of specific rapid response protocols and manuals have been developed for many species worldwide. Short-term evaluation and monitoring by the team will help to ensure that time is not wasted on a population that has spread beyond rapid response capabilities.

The funding required to manage or eradicate all the new IARs in Florida would be economically and politically prohibitive. In fact, the "tens" rule dictates that we do not need to thwart all invasions (Williamson 1996). Instead, we must develop a system to prioritize which species to eradicate in the face of limited funding. In prioritizing which introduced species to eradicate or manage, there is a stark contrast between small insectivores (e.g., anoles) and large conspicuous predators (boas, pythons, monitors, and other "charismatic megafauna") when determining which species will get attention and funding. In the past, Florida has mainly suffered introductions of small, seemingly insignificant species (e.g., small lizards). Small species such as the brown anole are often extremely abundant (Campbell and Echternacht 2003). Eradication of these species would be nearly impossible once firmly established, and any good cost-benefit analysis would contraindicate eradication unless it could be done very early in the invasion or involves small, isolated populations (T. Campbell, unpublished data). However, the playbook has changed dramatically with the establishment of large herbivores, omnivores, and carnivores. Because large introduced predators often inflict significant direct impacts on native species, the urgency to prevent their establishment is dire, and justification for their eradication is almost incalculable.

Species identity also matters, in that charismatic megafauna will generate very different responses, and some of the large, sexy species are more equal than others. For instance, people tend to like herbivorous green iguanas much more than carnivorous Nile monitors, and often defend iguanas vigorously; however, even their opinions about green iguanas changed when they became pests (Krysko et al. 2007). Location also matters, in that species in residential areas or expensive waterfront property (e.g., iguanas on Gasparilla Island) will get more attention and funding. It is also more difficult to educate the public, and have eyes on the situation, in rural areas (Argentine tegus in rural Tampa versus Nile monitors in residential Cape Coral). Responders must take these things into account when mounting an eradication attempt.

Finally, Rapid Response efforts should be defined as, and confined to, those species that can be eliminated in a short period of time, hopefully before successful breeding occurs. Initially, the new population is influenced by the Allee effect, where individuals occur at such low densities that they never or rarely find each other, and the population grows very slowly. This lag phase occurs before the rapid exponential or geometric growth phase, and is the best time to mount a rapid response effort from economical and logistical standpoints. A good threshold for Rapid Response may be to determine whether or not the population could be eradicated quickly (days or weeks) or will require a major effort (months or years).

**Post-Project Evaluation and Reporting**

Post-project assessment is critical to the success of any EDRR program (ISWG 2003). It is up to the entity administering the EDRR program to set a priori guidelines for program review. Program administrators must periodically assess the IAR-EDRR program in general and individual response efforts in particular. For instance, proper data collection and analysis will be required to index or estimate the size of an incipient population being diminished by trapping (Engeman 2005). Without such analyses, it will be difficult if not impossible to demonstrate whether or not RR efforts had any effect. Sufficient post-project assessment will provide accountability and generate a roadmap for improvement of future EDRR efforts.

**A Hopeful Case Study**

I recently embarked on a pilot EDRR program to keep Nile monitors from becoming established on Sanibel Island, a barrier island less than 10 km from the core Nile monitor population in Cape Coral. Sanibel Island is home to the J. N. Ding Darling National Wildlife Refuge, one of Florida's most important bird sanctuaries. The propagule pressure from Cape Coral is intense, a few Nile monitors have been observed on the island, and concern is mounting. I am currently working with the USFWS (W. Thomas, personal communication), the City of Sanibel (J. Zimomra, personal communication), and the Sanibel-Captiva Conservation Foundation (B. Smith, personal communication) to determine the extent of the problem and formulate an EDRR protocol to keep

the lizards from becoming established on the island. Hopefully, their program will be successful in thwarting the establishment of this species, will serve as a model for a larger EDRR program in Florida, and will even instill a bit of badly needed encouragement that we have the political will, funding, and expertise to thwart the establishment of the most devastating IARs in Florida.

## CONCLUSION

The extinction of the passenger pigeon (*Ectopistes migratorius*) shows that Americans can accomplish extraordinary eradication feats when they put their minds to it. In the face of limited resources, managers must carefully choose which species could and should be eradicated and which species are destined for management in perpetuity. While massive eradication efforts will be required to eliminate certain firmly established species, and management in perpetuity will be necessary for others with no hope of being eradicated, prevention is the only viable mechanism for stemming the tide of newly introduced amphibians and reptiles in Florida. What will it take to prevent additional non-native amphibians and reptiles from becoming established in Florida?

First, our attitude about IARs is important. The main solution is to stop studying them in such great detail (Simberloff 2003) and start (1) preventing them from becoming established, (2) eradicating recently established species before it becomes logistically and economically prohibitive, and (3) developing management tools to ameliorate the effects of species that cannot be eradicated. Although the "tens" rule predicts that few of those species that become established go on to be pests (Williamson 1996), we cannot yet predict their impacts in the future, so introduced species should be considered guilty until proven innocent (Van Driesche and Van Driesche 2001). EDRR is a logical extension of the GUPI philosophy in that we should eradicate first and ask questions later.

The most effective preventive policy will include a combination of multiple strategies. First, legislation restricting ownership and sale of certain problematic species is warranted. The first steps have already been taken, in the form of a law restricting sale and requiring a permit for ownership of certain large reptiles which goes into effect on January 1, 2008. Sufficient regulations of the pet industry, education of the public, and incentives for good behavior will be crucial. An example is the recently established Pet Amnesty Day, where pet owners are able to donate unwanted pets, for free and with no questions asked, in the hopes they will be adopted instead of released into the wild (S. Hardin, personal communication).

Given the new regulations, we must assume that some pet owners will avoid fees and regulations and simply release their pet into the wild. Others will arrive accidentally. Considering the myriad unrestricted pathways currently in place, the intense propagule pressure from the pet industry, the high fecundity of many amphibians and reptiles, and the excessive cost of dealing with them after they become established, an effective EDRR program will be necessary to thwart the establishment of incipient IAR populations in Florida and beyond. Such a program has applicability to species of invertebrates, fish, birds, and mammals that are popular in the pet trade. However, EDRR is but one element of a comprehensive prevention program.

Beyond EDRR, stakeholders must work proactively with the pet industry, herpetoculturists, and hobbyists to identify species that might become established in the future. Risk assessments based on which species are most popular and which species would be most likely to thrive in Florida's habitats and climate (using climate matching and GAP analysis) will enable specialists to predict the next problematic species.

Interest in establishing an EDRR program for IARs in Florida is gathering steam. Now we should determine who will fund and implement such a program in Florida and beyond, obtain funding, and hit the ground running. I hope this paper provides a solid foundation on which an effective EDRR program can be built.

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## LITERATURE CITED

BUTTERFIELD, B. P., W. E. MESHAKA, JR., AND C. GUYER. 1997. Nonindigenous amphibians and reptiles. Pages 123-138 *in* D. Simberloff, D. C. Schmitz, and T. C. Brown, editors. Strangers in paradise: impact and management of nonindigneous species in Florida. Island Press, Washington, D.C., USA.

CAMPBELL, T. S. 2000. Analyses of the effects of an exotic lizard (*Anolis sagrei*) on a native lizard (*Anolis carolinensis*) in Florida, using islands as experimental units. Dissertation, University of Tennessee, Knoxville, Tennessee, USA.

CAMPBELL, T. S. 2005. The impacts and management of the introduced Nile monitor (*Varanus niloticus*) in southwest Florida. Unpublished report to the Charlotte Harbor National Estuary Program, Port Charlotte, Florida, USA.

CAMPBELL, T. S. AND A. C. ECHTERNACHT. 2003. Introduced species as moving targets: body sizes of introduced lizards following experimental introductions and historical invasions. Biological Invasions 5:193-212.

CARR, A. F., JR. 1940. A contribution to the herpetology of Florida. University of Florida Biological Science Series Publication 3:1-118.

CARR, A. F., JR. AND C. J. GOIN. 1955. Guide to the reptiles, amphibians, and freshwater fishes of Florida. University of Florida Press, Gainesville, Florida, USA.

DALRYMPLE, G. H. 1994. Non-indigenous amphibians and reptiles in Florida. Pages 67-78 *in* D. C. Schmitz and T. C. Brown, editors. An Assessment of invasive non-indigenous species in Florida's public lands. Florida Department of Environmental Protection, Technical Report No. TSS-94-100, Tallahassee, Florida, USA.

DUELLMAN, W. E. AND A. SCHWARTZ. 1958. Amphibians and reptiles of south Florida. Bulletin of the Florida State Museum 3:181-324.

ENGE, K. M., K. L. KRYSKO, K. R. HANKINS, T. S. CAMPBELL, AND F. W. KING. 2004. Status of the Nile monitor (*Varanus niloticus*) in southwestern Florida. Southeastern Naturalist 3:571-582.

ENGEMAN, R. M. 2005. Indexing principles and a widely applicable paradigm for indexing animal populations. Wildlife Research 32:203-210.

FERRITER, A., B. DOREN, C. GOODYEAR, D. THAYER, J. BURCH, L. TOTH, M. BODLE, J. LANE, D. SCHMITZ, P. PRATT, S. SNOW, AND K. LANGELAND. 2006. Chapter 9: The status of nonindigenous species in the south Florida environment. South Florida Environmental Report, South Florida Water Management District, West Palm Beach, Florida, USA.

GENERAL ACCOUNTING OFFICE. 2001. Invasive species: obstacles hinder federal rapid response to growing threat. United States General Accounting Office Report GAO-01-724. Washington, D.C., USA.

INVASIVE SPECIES WORKING GROUP. 2003. Statewide Invasive Species Strategic Plan for Florida. Department of Environmental Protection, Tallahassee, Florida, USA.

KING, F. W., AND T. KRAKAUER. 1966. The exotic herpetofauna of southeast Florida. Quarterly Journal of the Florida Academy of Sciences 29:144-154.

KRAUS, F. AND E. W. CAMPBELL, III. 2002. Human-mediated escalation of a formerly eradicable problem: the invasion of Caribbean frogs in the Hawaiian Islands. Biological Invasions 4:327–332.

KRYSKO, K., W. F. KING, K. M. ENGE, AND A. REPPAS. 2003. Distribution of the introduced black spiny-tailed iguana (*Ctenosaura similis*) on the southwestern coast of Florida. Florida Scientist 66:74-79.

KRYSKO, K., K. M. ENGE, E. M. DONLAN, J. C. SEITZ, AND E. A. GOLDEN. 2007. Distribution, natural history, and impacts of the introduced green iguana (*Iguana iguana*) in Florida. Iguana 14:2-11.

LOCKWOOD, J. L., M. F. HOOPES, AND M. P. MARCHETTI. 2007. Invasion Ecology. Blackwell Publishing, Malden, Massachusetts, USA.

MESHAKA, W. E., JR., B. P. BUTTERFIELD, AND J. B. HAUGE. 2004. The exotic amphibians and reptiles of Florida. Krieger, Melbourne, Florida, USA.

MYERS, J. H., D. SIMBERLOFF, A. H. CURTIS, AND J. R. CAREY. 2000. Eradication revisited: dealing with exotic species. Trends in Ecology and Evolution 15:316-320.

NATIONAL INVASIVE SPECIES COUNCIL. 2001. Management plan: meeting the invasive species challenge. Washington, D.C., USA.

NATIONAL INVASIVE SPECIES COUNCIL. 2003. General guidelines for the establishment and evaluation of invasive species early detection and rapid response systems. Washington, D.C., USA.

PARKER, M. A. AND T. R. E. KEENEY. 2004. Aquatic Nuisance Species Task Force Report to Congress. Washington, D.C., USA.

PIMENTEL, D. 2002. Biological invasions: economic and environmental costs of alien plant, animal, and microbe species. CRC Press, Boca Raton, Florida, USA.

RICE, K. G., J. H. WADDLE, M. E. CROCKETT, AND A. D. DOVE. 2003. The effects of the Cuban treefrog (*Osteopilus septentrionalis*) on native treefrog populations within Everglades National Park. Pages 184-185 *in* A. E. Torres, A. L. Higer, H. S. Henkel, P. R. Mixson, J. R. Eggleston, T. L. Embry, and G. Clement, compilers. U.S. Geological Survey Open-File Report 03-54, Tallahassee, Florida, USA.

RODDA, G. H., Y. SAWAI, D. CHISZAR, AND H. TANAKA. 1999. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

SCHOENER, T. W. AND D. A. SPILLER. 1999. Indirect effects in an experimentally staged invasion by a major predator. American Naturalist 153:347-358.

SIMBERLOFF, D. 2003. How much information on population biology is needed to manage introduced species? Conservation Biology 17:83-92.

SIMBERLOFF, D. AND B. VON HOLLE. 1999. Positive interactions of nonindigenous species: invasional meltdown? Biological Invasions 1:21-32.

SIMBERLOFF, D., I. M. PARKER, AND P. N. WINDLE. 2005. Introduced species policy, management, and future research. Frontiers in Ecology and the Environment 3:12-20.

SIMBERLOFF, D., D. C. SCHMITZ, AND T. C. BROWN. 1997. Strangers in paradise: impact and management of nonindigneous species in Florida. Island Press, Washington, D.C., USA.

SMITH, H. M. AND A. J. KOHLER. 1978. A survey of herpetological introductions in the United States and Canada. Transactions of the Kansas Academy of Sciences 80:1-24.

VAN DRIESCHE, J. AND R. VAN DRIESCHE. 2001. Guilty until proven innocent: preventing nonnative species invasions. Conservation In Practice 2:8-17.

WILSON, L. D., AND L. PORRAS. 1983. The ecological impact of man on the south Florida herpetofauna. University of Kansas Museum of Natural History, Special Publication No. 9, Lawrence, Kansas, USA.

WILLIAMSON, M. 1996. Biological Invasions. Chapman & Hall, London, United Kingdom.

WORRALL, J. 2002. Review of systems for early detection and rapid response. Unpublished report to the National Invasive Species Council, Washington, D.C., USA.

# COQUI FROG RESEARCH AND MANAGEMENT EFFORTS IN HAWAI'I

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***Abstract:*** The coqui frog (*Eleutherodactylus coqui*) is native to Puerto Rico and was accidentally introduced to the State of Hawai'i through contaminated nursery products from the Caribbean. Since its introduction in the late 1980s, coqui have become widely dispersed and in some areas population densities have reached 55,000 frogs/ha. The coqui frog is a species of concern because individual frogs can produce sound levels of 80 decibels (at 0.5 m), which has caused sleep loss to residents and affected the quality of life in Hawai'i. Economic concerns in infested areas include diminished property values and sales, impacts on tourism, and decreased sales in the floriculture and nursery industry. In addition, research indicates that the coqui has potential ecological effects as they might predate on endangered invertebrates and shift nutrient cycling processes in native forests. Control efforts have focused on habitat modification and applying chemical solutions of either 16% citric acid or 3% hydrated lime. Hot water or vapor treatments of plants are also effective control methods. Eradication of the coqui frog is not considered attainable on the Island of Hawai'i and seems unlikely for Maui, but may be possible on the islands of Kaua'i and O'ahu.

***Key Words:*** amphibian, biological invasions, chemical application, *Eleutherodactylus coqui*, Hawaiian Islands, invasive species, island conservation.

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## INTRODUCTION

The State of Hawai'i lacks any native or endemic amphibian species due to the geographic isolation of the Hawaiian Archipelago. The nearest continent, North America, lies 3,200 km to the east and the nearest Pacific atolls are 1,600 km to the south. In spite of these distances, approximately 27 species of herpetofauna have been introduced to Hawai'i (Kraus 2003), with coqui frogs (*Eleutherodactylus coqui*), Jackson's chameleons (*Chamaeleo jacksonii xantholophus*), and brown anoles (*Anolis sagrei*) expected to have negative effects on native ecosystems (E. W. Campbell, Fish and Wildlife Service, personal communication). In particular, the coqui frog, has been a species of focus for both control and research efforts.

The coqui frog is endemic to Puerto Rico and is thought to have been introduced to Hawai'i in the late 1980s via the horticultural industry (Kraus et al. 1999). Although the first voucher specimen for the coqui frog was collected from the Island of Maui in 1997, population sizes and densities were already noticeably larger on the Island of Hawai'i (F. Duvall, DLNR-DOFAW, personal communication). The coqui frog has since become one of Hawai'i's most recognizable invasive species as populations have steadily encroached on human habitats.

Currently, the coqui frog occurs in all types of habitats from sea level to 1,300 m in elevation (H. Sin, DLNR-DOFAW, unpublished data) and can reach population densities up to 55,000 frogs/ha in some areas (N. Tuttle, Utah State University, unpublished data), with their highest densities occurring in wet locations on the windward sides of the islands (Kraus and Campbell 2002). Such high densities are partly due to high fecundity rates (4-6 clutches/yr, 16-41 eggs/clutch) (Townsend and Stewart 1994), direct development of eggs (no tadpole stage), lack of predators and competitors (Beard and Pitt 2006), abundant food sources, and generalist feeding behavior (Stewart and Woolbright 1996).

## ECOLOGICAL EFFECTS

Left unchecked, the coqui frog is expected to adversely affect native and endemic invertebrates and change certain ecological processes. At such high densities, this sit-and-wait, generalist predator can possibly consume greater than 400,000 invertebrates a night/ha, based on 55,000 frogs/ha with an average of 7.6 prey items consumed per

night per individual (Beard 2007a). This direct consumption has significant potential to reduce Hawai'i's invertebrate populations, which make up most of Hawai'i's endemic fauna (Eldredge and Miller 1995). One recent diet study in particular has provided evidence that the coqui frog predates on endemic invertebrates. Beard (2007a) suggests that Acarina (mites), Coleoptera (beetles), Collembola (springtails), Diptera (flies) and Gastropoda (snails) are the most vulnerable to predation and that each order comprised more than 1.5% of its total diet in a variety of habitats including nurseries, disturbed habitats, and natural areas.

The coqui frog may contribute to significant increases in nutrient recycling rates in the ecosystem from its excrement, which in turn may increase certain invasive plant growth and increase leaf litter decomposition rates (H. Sin et al. In Press). Increased leaf litter decomposition can in turn release additional nutrients for plant use, particularly invasive plants. Invasive plants in Hawai'i are usually able to utilize certain resources, such as nutrients, more efficiently than native plants (Pattison et al. 1998, Ostertag and Verville 2002). Specifically, in a small-scale enclosure experiment, the presence of the coqui frog reduced herbivory rates, increased certain nutrients in 'ōhi'a (*Metrosideros polymorpha*) leaf litter, increased 'ōhi'a leaf litter decomposition rates and increased the number of new leaves of strawberry guava (*Psidium cattleianum*) (H. Sin et al. In Press). These results are consistent with similar small- and large-scale experiments conducted in the coqui frog's home range in Puerto Rico (Beard et al. 2002, 2003).

The coqui frog also has been hypothesized to compete with native birds for food, the majority of these birds being insectivorous (Kraus et al. 1999). Some variables to consider when discussing potential competition of native birds and the coqui frog are: overlapping range, habitat, and common prey of these species. Currently, the majority of Hawaiian endemic birds are above 500 m in elevation (Stattersfield et al. 1998) and high densities of frogs are below this elevation (Beard and Pitt 2005, H. Sin, unpublished data). The current status of this condition suggests minimal competition between native birds and coqui frogs. However, there is a strong potential for coqui frogs to naturalize in these higher-elevation areas as they are capable of occurring in all types of habitats from sea level to 1,300 m in elevation (H. Sin, unpublished data).

There is also a degree of overlap in common prey items where native birds and coqui frogs co-occur. For example, the coqui frog has the potential to compete for food with such species as the 'elepaio (*Chasiempis spp.*), the 'i'iwi (*Vestiaria coccinea*), and particularly the endangered Hawaiian hoary bat (*Lasiurus cinereus semotus*) as they share common prey items and elevational range (Beard and Pitt 2005). Conversely, frogs might be a possible food source for native birds, though unlikely due to different activity periods of these groups. Furthermore, research indicates that the coqui frog is an opportunistic feeder foraging upon abundant prey items. This generalist feeding behavior may interrupt key ecosystem processes and reduce the availability of preferred invertebrates for native bird and bat populations.

The coqui frog may also increase populations of native bird predators, such as the black rat (*Rattus rattus*) and the small Indian mongoose (*Herpestes javanicus*), by serving as a food source (Kraus et al. 1999). However, a recent diet study of rodents (*R. rattus* and *R. exulans*), cane toads (*Bufo marinus*), and the small Indian mongoose (*H. javanicus*) on the Island of Hawai'i found that only the small Indian mongoose had consumed coqui frogs (Beard and Pitt 2006). These results are similar to those found in Puerto Rico, which has the same non-native rat, mongoose, and toad species (Beard 2007b). Further research is necessary to determine if coqui frogs are indeed bolstering mongoose populations. The coqui frog may also serve as a food source for other potentially devastating bird predators such as the brown tree snake (*Boiga irregularis*) (Beard and Pitt 2005).

## ECONOMIC EFFECTS

The coqui frog has adversely affected Hawai'i's economy, particularly through the nursery and real estate industries, due primarily to the volume and consistency of its vocalizations. The mating call of male coqui frogs, which typically begins at sunset and continues throughout the night, can reach sound levels of 80-90 decibels (at 0.5 m), exceeding the legislatively-established state health standard of 70 dBA (Hawai'i Revised Statutes § 324F-1) (Beard and Pitt 2005). Coqui frog vocalizations have resulted in sleep loss for both residents and tourists, which has decreased accommodation revenues in some areas and has negatively affected real estate markets (Kaiser and Burnett 2006). Hotels have reported complaints of coqui frogs and some visitors report that they have

changed, or are planning to change, their travel plans to avoid coqui frogs in the future (W. C. Pitt, USDA-APHIS-WS-NWRC, personal communication). Residents in heavily populated areas report that the introduction of coqui frogs to their community has negatively affected their quality of life and may influence their willingness to live, or continue to live, in impacted areas. Effects on hotels, resorts, and displacement of residents are largely anecdotal and more research is needed to quantify these impacts.

Property owners on the Island of Hawai'i have also felt the economic effects of coqui frogs and are currently required to disclose if frogs are present on the property before selling their property or homes, similar to termites; which has resulted in an average of 0.16% loss of real estate value per sale (Kaiser and Burnett 2006). Though this percentage appears small on a per household basis, the total direct damage to property values for all homes in the State of Hawai'i is conservatively estimated at $208.8 million (Kaiser and Burnett 2006).

The nursery industry has been particularly affected by the presence of coqui frogs. The State of Hawai'i has designated the coqui frog as a "pest" and "injurious wildlife" under Hawai'i Revised Statutes (HRS) 141-3 and 124-13, respectively. These laws establish provisions for both cooperative and mandatory control of the coqui frog, and make it illegal to release, transport or export coqui frogs. These requirements have resulted in rejection of nursery goods at ports of entry, lost time in shipping, added labor costs for nurseries, and more stringent preventative/ treatment measures for invasions of coqui frogs. Indirect costs to nurseries and plant providers have increased as consumers have become more selective purchasers. Although no citations have been issued to businesses or individuals, the Hawai'i Department of Agriculture's (HDOA) website (www.hawaii.gov/hdoa) states: Any person or organization who intentionally transports, harbors or imports with the intent to propagate, sell, or release the coqui is in violation of State law and may be charged with a class C felony and subject to a minimum fine of $50,000 and a maximum fine of $200,000, plus 3 years in prison.

Currently, the HDOA and other public and private entities are working on both mandatory and voluntary "Coqui-Free" certifications for nurseries to minimize the possibility of inadvertent distribution. These programs are also intended to help reassure apprehensive consumers and build confidence that the inadvertent spread of coqui frogs is being limited.

## CONTROL METHODS

A number of methods are currently being used to control the proliferation of coqui frogs in the State of Hawai'i. Both chemical and non-chemical control methods continue to emerge and are evaluated on a case-by-case basis by researchers from the HDOA, University of Hawai'i's College of Tropical Agriculture and Human Resources (CTAHR), and USDA-APHIS-NWRC in Hilo. Given the overwhelming abundance of coqui frogs on some private properties, unregulated independent trials, misuse of chemicals, and experimentation with control methods occur. Often new approaches and successful strategies are shared via e-mail lists, informal conversations, and at community and professional meetings throughout the state. Federal, state and county organizations work hard to help facilitate cooperative control initiatives and to ensure that chemical labels are being followed.

### Chemical Control Methods

The most widely used chemical methods for controlling coqui frogs are to apply either 16% citric acid or 3% hydrated lime solutions to an infested area. Both chemicals have been shown to be effective toxicants for frogs while also reducing egg hatch rates. However, direct application of solutions onto frogs and eggs are necessary to be effective; thus, several applications may be needed to ensure that all frogs and eggs are eliminated (Beard and Pitt 2005). Historically, laboratory and field studies demonstrated that a 2% caffeine solution was an effective frog toxicant and did not have significant harmful effects on non-target species (Campbell 2001a, b; Pitt and Sin 2004a). However, caffeine is no longer registered for use as a frog toxicant and there are no future plans for registration given extensive testing requirements associated with concerns over potential human health effects. USDA-APHIS-WS-NWRC has also reviewed a long list of additional chemicals, both non-restricted and restricted use, but additional products have yet to emerge that are both lethal and cost-effective.

A 16 % citric acid solution was found to be effective at controlling frogs in laboratory and field settings (Pitt and Sin 2004b, c). Citric acid is classified as a "minimum risk" pesticide and is exempt from requirements of the Federal

Insecticide, Fungicide, and Rodenticide Act (FIFRA) 40 CFR 152.25. Subsequently, this product was put to use in the field after it was found to be effective on frogs. Citric acid does not affect most non-target arthropods (Pitt and Sin 2004c). However, citric acid may result in some phytotoxic effects and may not be appropriate for all greenhouse plants (Pitt and Sin 2004d). These potential phytotoxic effects, along with the relatively high cost (~$1.00/gallon), make this product undesirable in some situations. Some variation in the percent of active ingredient and addition of pyrethrins has occurred with citric acid use in an attempt to improve efficacy and minimize phytotoxic effects (Scott Williamson, DLNR, personal communication).

On April 26, 2005, a 3% hydrated lime solution was registered for use by HDOA under a Section 18 quarantine exemption from the US Environmental Protection Agency. This registration will expire on April 26, 2008. The 3% hydrated lime solution has proven to be an effective frog toxicant in both laboratory and field settings and has minimal non-target effects on arthropods (Pitt and Doratt 2005). Laboratory tests revealed that higher concentrations of hydrated lime (6%) will be more effective, especially when used as a soil drench. However, the label for hydrated lime currently restricts the concentration to 3%. Hydrated lime may be corrosive and results in white residue on plants and surfaces. As a result, hydrated lime is often not a preferred method in the floriculture industry where slight blemishes on plants make them undesirable. Hydrated lime does have a lower cost (~$0.06/gallon) than citric acid and is often more attractive to homeowners because costs tend to rapidly accumulate when treating large areas. However, there are more safety and environmental concerns with using hydrated lime than with citric acid, such as the caustic effects to skin by hydrated lime exposure.

**Chemical Application**

The most common method for applying citric acid and hydrated lime is through comprehensive sprays of infested areas using 100- to 400-gallon agricultural sprayers with agitators. Hose size varies from 0.3 to 1.5 inches, and sprayer capacities range from small 1-gallon sprayers to 400-gallon spray tanks. The most effective applications are conducted during early evening hours when frogs are emerging from retreat sites. Multiple applications of selected chemicals are necessary to achieve control in areas with high densities of frogs. Water cannons and large volume irrigation equipment techniques are also being evaluated as application tools.

Aerial application of citric acid has been explored as an alternative method when it is not feasible to conduct ground operations due to challenging terrain or inaccessible frog populations. The label for hydrated lime does not allow aerial application. This airborne approach is similar to fire fighting methods where a helicopter uses a bambi bucket (100-gallon capacity) to drop citric acid solution instead of water. Common mixing procedures consist of using 400 gallon agriculture sprayers to prepare the chemical and transfer it to the large-volume dipping tank.

This application procedure has comparable costs on a per acre basis with ground operations largely because more area can be treated in a shorter period of time. Additionally, aerial application has been found to reduce densities of frogs (N. Tuttle, Utah State University, unpublished data). However, after several trials on the Island of Hawai'i this approach did not appear to have comparable control efficacy to ground-based operations in areas that are dry and have complex a'a substrates, which provides numerous retreat sites for the coqui frog (H. Sin, unpublished data). This reduced efficacy, compared with ground operations, maybe in part due to time of application (daytime) and substrate. Thus, this application technique may prove to be more effective in areas that have a greater soil substrate and where chemicals can have easier access to the frogs.

There have been no observed or reported harmful non-target effects from application of citric acid, though there is concern about potential effects on the endangered Hawaiian hoary bat. Preliminary exposure tests by the USDA/NWRC using bat effigies suggested no negative impacts during an aerial application (Pitt and Swift 2006). To further address concerns about citric acid exposure to hoary bats, USDA/NWRC proposes to conduct an LD50 study using the big brown bat (*Eptesicus fuscus*) as a surrogate species (W. C. Pitt, USDA-APHIS-WS-NWRC, personal communication).

To determine efficacy of chemical treatment, follow-up monitoring is necessary; this can be performed by both listening for calling males and conducting line transect searches for frogs. Additionally, there is continuing research and development for efficacy of treatment using sound decibel readings (M. Warrington et al., University of Hawai'i at Hilo, unpublished data). Preliminarily

data suggests that once a frog population reaches a certain density sound levels plateau even if the population continues to grow.

**Non-Chemical Control Methods**

Several non-chemical control methods are available for plant quarantine, the nursery industry, and homeowners. These methods include hand capture, habitat management, hot water treatment, and placement of coqui barriers. Coqui frog traps, in a variety of forms, have been tested. Some data suggests that small PVC pipes placed one to two-feet off of the ground can trap coqui frogs (A. Hara, CTAHR, unpublished data). However, no traps have proven to be effective on a broad scale to date.

In areas with few frogs, hand capture can be accomplished with an observer slowly approaching the frog at night and quickly grabbing or placing a plastic tube over the individual. Locating females and juveniles in the field can be difficult though because only adult males vocalize.

Habitat management through physical removal of dense vegetation reduces the carrying capacity of the site for frogs by making the area less hospitable for them. The coqui frog prefers dense vegetation, which provides more food, cover, and higher moisture levels compared with a more open area, such as a manicured lawn (W. J. Mautz, University of Hawai'i at Hilo, unpublished data). Additionally, more open areas appear to greatly improve the efficacy of chemical applications (A. Radford and H. Sin, unpublished data).

Treating plants with hot water or hot water vapor has been shown to be effective in removing frogs and eggs. Research indicates that treating plants at 113°F for five minutes can kill frogs and eggs with minimal damage to most common nursery plants (A. Hara, CTAHR, unpublished data). Furthermore, if temperatures remain constant for longer periods, or the temperature is raised to 120°F, other quarantine pests, such as scales and mealybugs, can also be removed. Certain plants, especially flowering ones, may become more susceptible to damage from raised water temperatures (A. Hara, CTAHR, unpublished data). These findings, coupled with subsequent field trials of hot water treatment facilities, have yielded positive results. Plans are in progress to install treatment facilities at many of Hawai'i's ports of entry.

Another preventive technique being explored is the use of a coqui barrier or fence. Designed by the HDOA, the fence restricts coqui frog movements from infested areas to frog-free areas. The fence consists of a two- to three-foot high fine insect mesh supported by posts spaced several feet apart. The barrier has a 90° lip at the top that extends approximately one foot toward the infested area. Laboratory tests have demonstrated that a frog can climb up the vertical portion but falls off the horizontal lip due to lack of traction from its toe pads (Kyle Onuma, HDOA, personal communication). Additionally, the height of the fence adequately discourages the frog from jumping over the structure. However, the barrier needs constant maintenance to ensure there is no overhanging vegetation and to ensure the structure remains intact. This technique is more applicable to small areas such as greenhouses rather than larger natural areas.

**Biological Control**

Biological controls that have been considered include chytrid fungus (*Batrachochytrium dendrobatidis*) and internal parasites. Chytrid fungus is known to cause a lethal amphibian disease called chytridiomycosis and has affected amphibians in Australia, South America, Central America and the United States (Berger et al. 1999). This fungus is a water born pathogen that will not likely affect the coqui frog as it does not have a tadpole stage. Additionally, chytrid fungus has already been found in low percentages of frogs in Hawai'i and does not appear to affect coqui populations (Beard and O'neill 2005). Internal parasites have been found during examination of coqui frogs collected in Puerto Rico. Further research will investigate the effects of these parasites on the health of coqui frogs and potential effects on non-target organisms (S. Marr, CTAHR, unpublished data).

## QUARANTINE

Continued inter- and intra-island movement of coqui frogs through the nursery trade and other inadvertent vectors (e.g., movement of vehicles, household goods, or green waste) seriously undermines efforts to control the spread of the coqui frog. Inter-island inspection of nursery plants is being conducted by the HDOA for the presence of frogs. Nurseries that are known to have frogs are required to treat plants before shipping to other islands and may be rejected at ports of entry if products are found to be infested. Both nurseries and plant material are also subject to random HDOA inspections. A nursery grant program is underway to provide matching funds for individual

nurseries to develop on-site hot water treatment facilities.

## PUBLIC OUTREACH AND EDUCATION

Public involvement is crucial to the discovery of new frog populations. Hotlines on all islands allow the public to report frog locations. Residents in many areas are the only source of information regarding the current level of infestation. Public support is necessary for control because a large portion of infestations occur on private property. Consequently, public education and awareness building are necessary ingredients to facilitate reporting, identification, and control of coqui frogs. Public service announcements, brochures, websites, newsletters, and presentations are some of the methods used to reach to the public on this issue. On the Island of Hawai'i, residents are largely responsible for their own control efforts. Thus, training the public on control strategies, personal safety when handling chemicals and awareness of available resources are extremely important. This training occurs at some level on all affected islands. Additionally, there have been instances where the public has intentionally moved frogs or have hampered control efforts, which further highlight the need for public outreach and education to prevent such cases.

## ISLAND-SPECIFIC MANAGEMENT STRATEGIES

Management goals are island-dependent since each island has various levels of coqui frog infestations with funding availability and the infestation level influencing control efforts. However, common strategies do exist and include: containing or eradicating known populations; protecting high-value areas; rapidly responding to new reports; removing incipient populations; preventing reintroduction; continuous monitoring of infested and treated sites; collecting data and appropriately documenting efforts; disseminating information and resources to the public; and developing situation-dependent plans and protocols. A broader statewide plan is being developed which may assist with on-the-ground efforts, provide clear identification of responsibilities and duties, and improve consistency in data collection and reporting. These elements will help ensure a coordinated response to new reports of coqui frogs. Public reporting also has been a key element in successful statewide management of coqui frogs. As a result of thousands of reports, the current infested areas are relatively well known.

For management purposes in the State of Hawai'i, incipient populations are generally considered to be four or less vocalizing males heard in one location or in the surrounding area. A naturalized population is generally considered to be five or more vocalizing male coqui heard in one location or in the surrounding area. A naturalized population is declared eradicated if no frogs have been detected for a year from the date the last vocalizing coqui frog was heard, based on surveys conducted periodically post-treatment.

### Hawai'i County

Anecdotal evidence suggests that coqui frogs were first detected on the Island of Hawai'i in the late 1980s. The strategy, where coqui populations are numerous and dense, is to control and minimize distribution in priority areas. However, incipient populations and occurrences of single coqui frogs are also targeted for eradication as they become known. Areas of priority are high-value natural areas, nurseries, residential areas, parks, waste transfer stations, and other potential distribution centers. The Island of Hawai'i has the largest number of naturalized populations and largest total infestation, with more than 20,000 infested acres, concentrated on the northeast, or Hilo, side of the island (Figure 1). All levels of government are involved in coqui management on the Island of Hawai'i and as a result the Coqui Frog Working Group (CFWG) was initiated as a cooperative entity to help coordinate management strategies. Groups involved with the CFWG include the Big Island Association of Nurserymen, County of Hawai'i, University of Hawai'i, CTAHR, USDA-APHIS-Wildlife Services, USDA-APHIS-WS-NWRC, Department of Land and Natural Resources, HDOA, and the Big Island Invasive Species Committee. Management approaches include all available techniques and the majority of coqui research and development of new control methods occurs on the island. State and county sprayer loan programs, community grant programs, and matching chemical cost programs, are in place to assist and encourage affected residents to take action.

### Maui County

Coqui frogs were first detected on Maui in 1997, on Lāna'i in 2002, and on Moloka'i in 2001. No coqui frogs have been observed on Kaho'olawe. Since 1997, thirteen naturalized population centers




**Figure 1.** Current locations and control efforts of the coqui frog (*Eleutherodactylus coqui*) on the Island of Hawai'i. Given the large scale of the map some points were consolidated into one point. Furthermore, most of the locations are based on hotline calls and are not necessarily indicative of coqui frog densities.

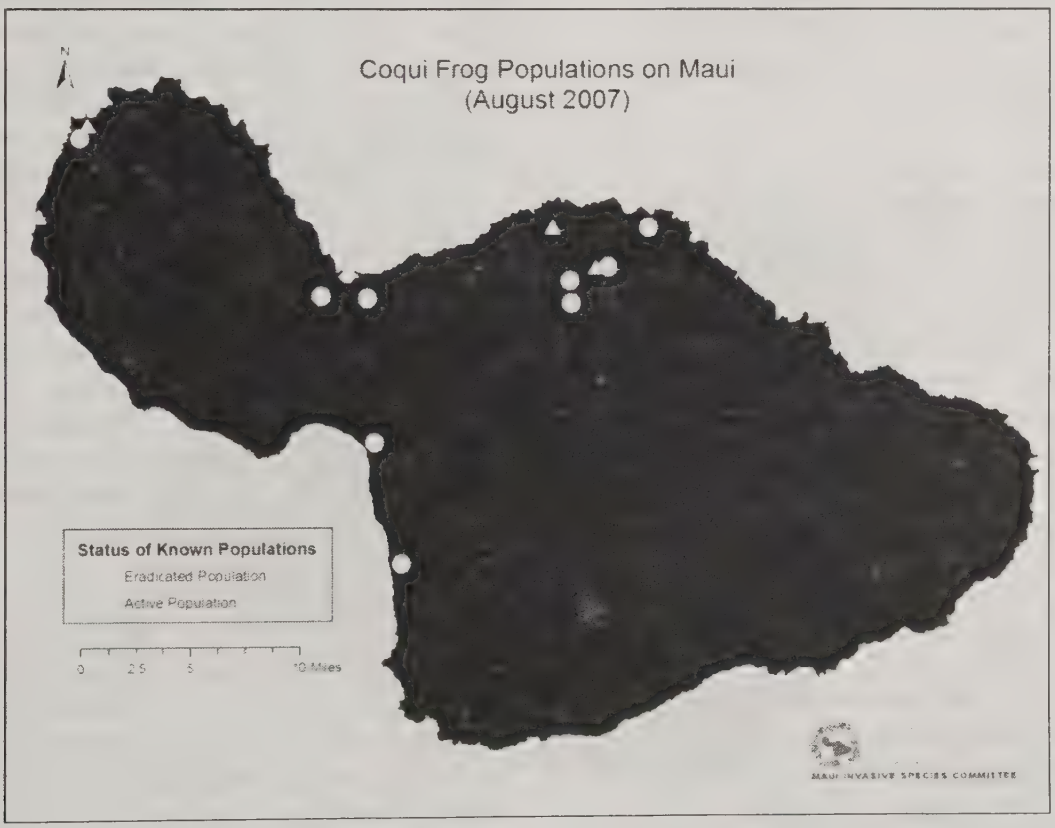



**Figure 2.** Current naturalized and eradicated populations of the coqui frog (*Eleutherodactylus coqui*) on the Island of Maui.

have been identified on Maui covering 161 acres (Figure 2). No coqui frog populations are known to exist on Lāna'i or Moloka'i and the four occurrences of calling males have been removed from these islands through a combination of hand capture techniques, and spraying of citric acid.

Work has begun in all known infested areas on Maui and eradication seems likely for all locations with the exception of a heavily-infested gulch. Four of the thirteen coqui frog populations are considered eradicated. Eight of thirteen populations are considered contained and appear to be headed toward eradication. The thirteenth population center includes a 76-acre gulch that has proven difficult to manage due to inaccessible terrain. Maui managers are hopeful that coqui frogs can be contained or eradicated from the gulch. Long-term strategy will focus on monitoring historic population centers and continuing to rapidly respond to new reports and introductions. The Maui Invasive Species Committee (MISC) spearheads control work on Maui with a supplemental sprayer loan program sponsored by the HDOA. Management approaches on Maui include citric acid applications in infested areas, distribution of citric acid to affected residents, habitat modification and removal, hand capture, and systematic follow-up.

**O'ahu and Kaua'i County**

Coqui frogs were first detected on O'ahu in 1998 and on Kaua'i in 2000. Coqui frogs have never been observed on Ni'ihau. O'ahu and Kaua'i each have one population center with each island headed toward a coqui-free status. On O'ahu, the first observation of a coqui frog was made in 1998. Since then, there have been confirmed reports of coqui frogs scattered across the island. Most individuals or small numbers of coqui frogs have been removed and work continues at the one known population at the Schofield Barracks in Wahiawa. Wahiawa is located in central O'ahu and no coqui frogs have been reported in the 12-acre area since November of 2006 (Figure 3). Work on O'ahu occurs under the oversight of a cooperative Coqui Working Group with participation by multiple agencies. Participant agencies include the Department of Public Works, US Army Environmental, US Fish and Wildlife Service, City and County of Honolulu, DLNR-DOFAW, HDOA, and the O'ahu Invasive Species Committee (OISC). Suppression efforts include citric acid applications in previously infested areas and in areas with incipient populations, maintenance of previously reduced habitat, and systematic follow-up.

Kaua'i's population of coqui frogs is located in Lawai, on the southwest corner of the island and covers approximately 15 acres (Figure 4). Control work is spearheaded by the Kaua'i Invasive Species Committee (KISC), with cooperative assistance from the HDOA and Kaua'i County. Management approaches rely on both citric acid and hydrated lime applications in infested areas, habitat modification and removal, hand capture, and systematic follow-up.

## CONCLUSION AND MANAGEMENT IMPLICATIONS

The coqui frog is a highly adaptable, prolific species that has demonstrated an ability to spread rapidly, if not controlled, in Hawai'i. It poses serious direct and indirect threats to Hawai'i's unique ecosystems given its ability to attain some of the highest densities ever observed for terrestrial amphibian populations (Stewart and Woolbright 1996). The presence of coqui frogs poses major economic and quality of life concerns for local residents and Hawai'i's tourist-based economy. Control efforts in Hawai'i have cost millions of dollars to date. Although remarkable progress has been made in some areas, without improved inter- and intra-island quarantine measures, it is expected that coqui frogs will continue to be reintroduced to all islands, including areas where the frogs have already been eradicated.

More research is needed to better understand and quantify the ecological and economic impacts of the coqui frog. Specifically, research to find if coqui frogs can indeed bolster mongoose populations and to quantify if the frogs can compete with native birds and the native bat for food. Additional research to quantify the coqui frog's effects on the nursery and tourist industries would also be helpful.

Eradication of the coqui frog is not considered attainable on the Island of Hawai'i, and seems unlikely for Maui, but may be possible on the islands of Kaua'i and O'ahu. Overall, coqui populations are continuously expanding on the Island of Hawai'i, with some levels of control, but all other islands are experiencing declining coqui populations due to management practices. Current control techniques and methods must continue to evolve to improve control efficacy, especially given the scale of the infestation on the Island of Hawai'i and particularly with regard to finding more cost-

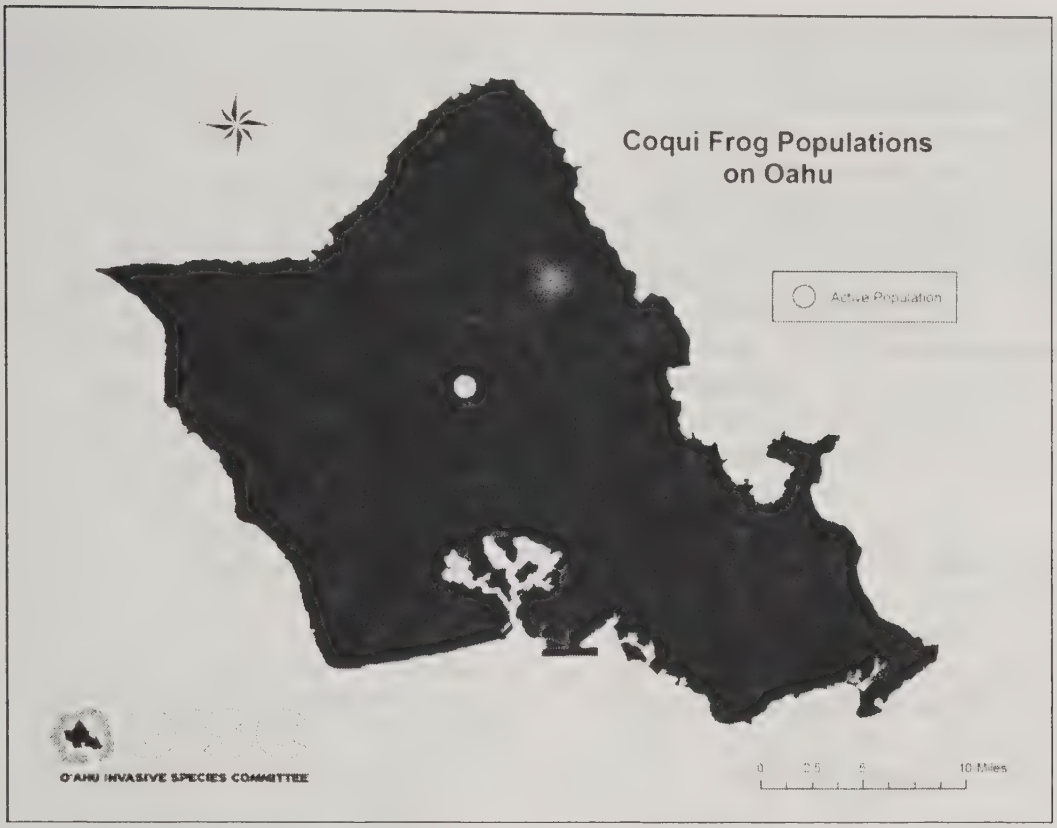



**Figure 3.** Current naturalized populations of the coqui frog (*Eleutherodactylus coqui*) on the Island of O'ahu.

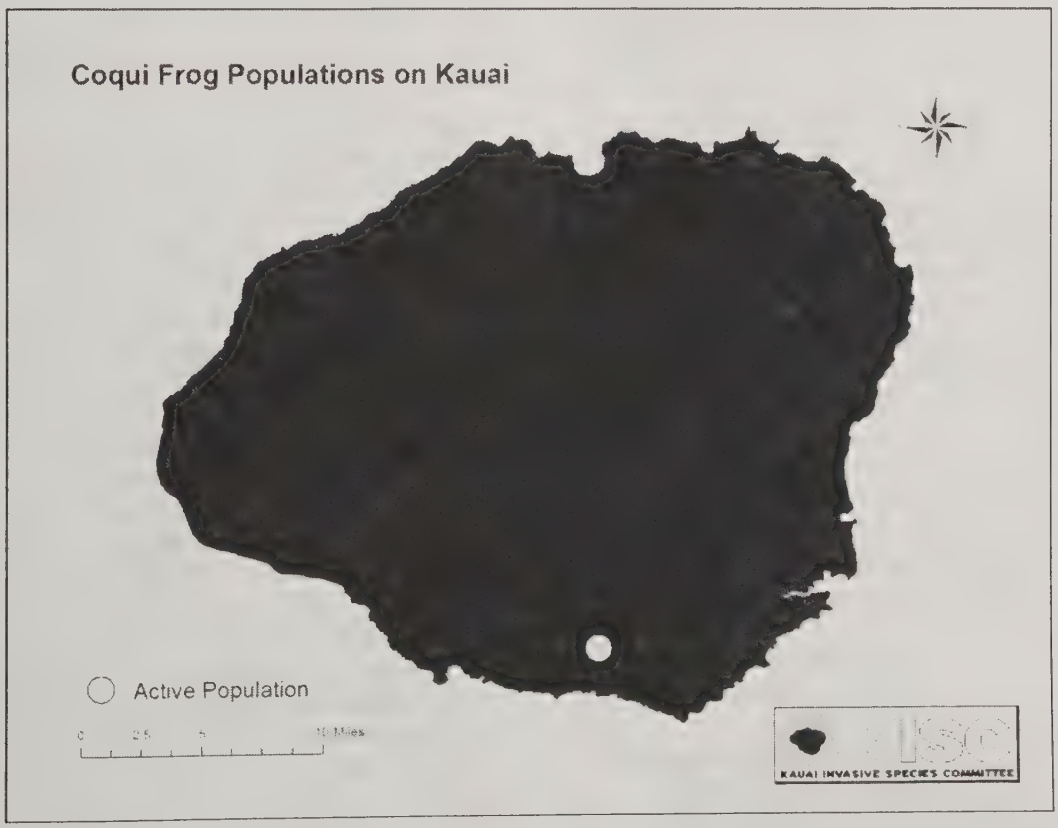



**Figure 4.** Current naturalized populations of the coqui frog (*Eleutherodactylus coqui*) on the Island of Kaua'i.

effective frog toxicants. If coqui frog populations continue to grow and/or move within and between islands, managers will rely more than ever on public reporting. In order to gain or keep public support, every tool at the disposal of respondents will be needed to ensure that appropriate control ensues. The full extent of the ecological and economic impacts of the coqui frog are still being determined, but the costly lessons from this invasion underscore the importance of adequate quarantine and enforcement of existing regulations, timely and relevant research, effective response and control techniques, and unified public support and involvement.

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## LITERATURE CITED

BEARD, K. H. 2007a. Diet of the invasive frog, *Eleutherodactylus coqui*, in Hawaii. Copeia 2007:281-291.

BEARD, K. H. 2007b. Coqui frogs as predators and prey in Hawai'i. Newsletter of the Maui Invasive Species Committee. Spring 2007:6-7.

BEARD, K. H., A. K. ESCHTRUTH, K. A. VOGT, D. J. VOGT, AND F. N. SCATENA. 2003. The effects of the frog *Eleutherodactylus coqui* on invertebrates and ecosystem processes at two scales in the Luquillo Experimental Forest, Puerto Rico. Journal of Tropical Ecology 19:607-617.

BEARD, K. H. AND E. M. O'NEILL. 2005. Infection of an invasive frog *Eleutherodactylus coqui* by the chytrid fungus *Batrachochytrium dendrobatidis* in Hawaii. Biological Conservation 126:591-595.

BEARD, K. H. AND W. C. PITT. 2005. Potential consequences of the coqui frog invasion in Hawaii. Diversity and Distributions 11:427-433.

BEARD, K. H. AND W. C. PITT. 2006. Potential predators of an invasive frog (*Eleutherodactylus coqui*) in Hawaiian forests. Journal of Tropical Ecology 22:345-347.

BEARD, K. H., K. A. VOGT, AND A. KULMATISKI. 2002. Top-down effects of a terrestrial frog on nutrient dynamics. Oecologia 133:583-593.

BERGER, L., R. SPEAR, AND A. KENT. 1999. Diagnosis of chytridiomycosis in amphibians by histologic examination. Zoos' Print Journal 15:184-190.

CAMPBELL, E. W. 2001a. Dermal toxicity of selected agricultural pesticides, pharmaceutical products, and household chemicals to introduced *Eleutherodactylus* frogs. Report submitted to the Maui Invasive Species Committee and the Hawaii Department of Land and Natural Resources.

CAMPBELL, E. W. 2001b. Field efficacy trials of the directed spray application of caffeine solutions for controlling introduced *Eleutherodactylus* frog in floriculture and nursery crops in Hawaii. Report submitted to the Hawaii Department of Agriculture and the USDI - Fish and Wildlife Service.

ELDREDGE, L. G., AND S. E. MILLER. 1995. How many species are there in Hawaii? Bishop Museum Occasional Papers 41:3-18.

KAISER, B. A., AND K. BURNETT. 2006. Economic impacts of *E. coqui* frogs in Hawaii. Interdisciplinary Environmental Review 8:1-11.

KRAUS, F., E. W. CAMPBELL, A. ALLISON, AND T. PRATT. 1999. *Eleutherodactylus* frog introductions to Hawaii. Herpetological Review 30:21-25.

KRAUS, F., AND E. W. CAMPBELL. 2002. Human-mediated escalation of a formerly eradicable problem: the invasion of Caribbean frogs in the Hawaiian Islands. Biological Invasions 4:327-332.

KRAUS, F. 2003. Invasion pathways of terrestrial vertebrates. Pages 68-92 *in* G. M. Ruiz, J. T. Carlton, editors. Invasive species: vectors and management strategies. Washington, D.C., USA.

OSTERTAG, R., AND J. H. VERVILLE. 2002. Fertilization with nitrogen and phosphorus increases abundance of non-native species in Hawaiian montane forests. Plant Ecology 162:77-90.

PATTISON, R. R., G. GOLDSTEIN, AND A. ARES. 1998. Growth, biomass allocation and photosynthesis of invasive and native Hawaiian rainforest species. Oecologia 117:449-459.

PITT, W. C., AND H. SIN. 2004a. Invertebrate non-target hazard assessment of caffeine application for control of *Eleutherodactylus* frogs. USDA, APHIS, WS, NWRC. Hilo, Hawaii, USA.

PITT, W. C., AND H. SIN. 2004b. Dermal toxicity of citric acid based pesticides to introduced *Eleutherodactylus* frogs in Hawaii. USDA, APHIS, WS, NWRC. Hilo, Hawaii, USA.

PITT, W. C., AND H. SIN. 2004c. Field efficacy and invertebrate non-target hazard assessment of citric acid spray application for control of *Eleutherodactylus* frogs in Hawaii. USDA, APHIS, WS, NWRC. Hilo, Hawaii, USA.

PITT, W. C., AND H. SIN. 2004d. Testing citric acid use on plants. Landscape Hawaii July/August 5/12.

PITT, W. C., AND R. E. DORATT. 2005. Efficacy of hydrated lime on *Eleutherodactylus coqui* and an operational field-application assessment on the effects on non-target invertebrate organisms. USDA, APHIS, WS, NWRC. Hilo, Hawaii, USA.

PITT W. C., AND R. SWIFT. 2006. Potential exposure of Hawaiian hoary bats to citric acid during aerial sprays. Report submitted to Hawaii Department of Land Natural Resources - Division of Forestry and Wildlife.

SIN H., K. H. BEARD, AND W.C. PITT. In Press. An invasive frog, *Eleutherodactylus coqui*, increases new leaf production and leaf litter decomposition rates through nutrient cycling in Hawaii. Biological Invasions.

STATTERSFIELD, A .J., M. J. CROSBY, A. J. LONG, AND D. C. WEGE. 1998. Endemic bird areas of the world: priorities for biodiversity conservation. BirdLife International, Cambridge, United Kingdom.

STEWART, M. M., AND L. L.WOOLBRIGHT. 1996. Amphibians. Pages 273-320 *in* D. P. Reagan, and R. B. Waide, editors. The food web of a tropical rain forest. University of Chicago Press, Chicago, Illinois, USA.

TOWNSEND, D. S., AND M. M. STEWART. 1994. Reproductive ecology of the Puerto Rican frog *Eleutherodactylus coqui*. Journal of Herpetology 28:34-40.

# IMPACTS OF INTRODUCED SPECIES ON AN ISLAND ECOSYSTEM: NON-NATIVE REPTILES AND AMPHIBIANS IN THE US VIRGIN ISLANDS

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***Abstract:*** Island ecosystems are highly sensitive to the impacts of introduced species. Non-native invasive snakes, lizards, and amphibians can introduce diseases into native populations and have other deleterious effects through predation, competition, and habitat manipulation. The United States (US) Virgin Islands are situated on the Puerto Rican Shelf in the Caribbean Sea and have a long history of human impacts and species introductions. Two species, the green iguana (*Iguana iguana*) and the red-legged tortoise (*Geochelone carbonaria*) were historically introduced and have become naturalized with little apparent impact to the local ecosystem. Recent years, however, have seen the introduction of several highly invasive species that have proved to have severe impacts when introduced elsewhere, specifically the Cuban treefrog (*Osteopilus septentrionalis*), cane toad (*Bufo marinus*), and the red-eared slider (*Trachemys scripta*). The distribution of non-native herpetofauna has been documented on all main islands of the US Virgin Islands and potential routes for introduction and dispersal have been identified. Additional species are still being identified, having arrived via cargo shipments from other Caribbean ports.

***Key Words:*** colonization, dispersal, herpetofauna, island conservation, invasive species, US Virgin Islands.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Situated near the eastern terminus of the Greater Antillean chain of islands in the northern Caribbean Sea, the United States (US) Virgin Islands (USVI) comprise four major inhabited islands and more than 50 smaller offshore cays. St. Thomas, St. John, and Water Island are the three main inhabited northern islands, located on the Puerto Rican Shelf to the east of Puerto Rico, while St. Croix is on a separate shelf to the south. The islands are mostly volcanic in origin, with steep slopes and irregular coastlines, surrounded by coral reefs and seagrass beds. The terrain is characterized by these steep hillsides with thin soils and a low permeability of underlying rock dominated by dry tropical forest. The highest elevation is on St. Thomas (474 m), which has an area of 74 $km^2$ and is the most densely populated of the islands (USDA-NRCS 1998). St. Croix (217 $km^2$) is more rural with extensive agriculture, and St. John (50 $km^2$) enjoys the protection of the US National Park Service over approximately two-thirds of its area. Combined, all offshore cays comprise about 3% (12 $km^2$) of the territory's area, most of which are designated wildlife refuges (Platenberg et al. 2005). The mild subtropical climate is influenced by easterly trade winds with little variation in average seasonal temperatures (Rogoziński 1999).

Historically, the Virgin Islands have been important for human colonization and settlement. The first inhabitants are dated to about 1500 BC and limited agriculture was in practice by Pre-Taino Indians by about 600 AD, with a heavy reliance on marine resources for food (Brewer 2005). They likely brought familiar and reliable food sources (e.g., tortoises and iguanas) from island to island as they made their way along the Antillean chain. By the time Columbus "discovered" the islands in 1492, the Island Caribs had already become established on St. Croix and throughout the region (Rogoziński 1999). The ensuing centuries saw the elimination of the local inhabitants with an expansion of European colonists and extensive deforestation to make way for sugar cane plantations. Control of the Virgin Islands passed from the Danish to the US in 1917, who established a naval base active through World War II. Tourism took off in the 1950s, and the USVI has become a major tourist destination noted for tax-free shopping and marine recreation.

The demands for space by a rapidly growing population of over 100,000 people have resulted in extensive loss and degradation of natural ecosystems in the USVI, especially on densely populated St. Thomas (Platenberg et al. 2005). Sprawling residential communities and commercial

centers have replaced or fragmented much of the native forest. Resorts, condominiums, and marinas have been constructed on coastal wetlands and marine recreational activities have damaged fragile mangrove swamps, coral reefs, and seagrass beds. Human development has led to increased pollution, particularly non-point source pollution that ultimately contaminates wetlands and marine environments. The introduction of exotic and feral animals and the spread of exotic landscaping plants have had a major impact on wildlife and habitats. Moreover, the natural ecosystems are subject to the effects of short- and long-term wet and dry climatic cycles, and to periodic disturbances from hurricanes. Most of the land within the USVI is divided into small privately-owned parcels that make survey and implementation of conservation measures difficult.

The Spanish and subsequent European colonizers introduced plants, animals, and pathogens from Europe, Africa, and South America (Rogoziński 1999); the most detrimental to native wildlife being the ship rat (*Rattus rattus*) and the small Indian mongoose (*Herpestes javanicus*). Some species arrived unnoticed during this early period of international trade, such as the house gecko (*Hemidactylus mabouia*). Other widespread species undoubtedly took advantage of the increased dispersal opportunities to colonize new areas within the region. Recent improvements in shipping, along with increasing human populations and demands for non-regional produce and landscaping and building materials, have enabled the dispersal of another group of opportunistic invaders (e.g., corn snake, *Elaphe guttata*, and Cuban treefrog, *Osteopilus septentrionalis*).

## HERPETOFAUNAL INTRODUCTIONS

A full 25% of the extant native terrestrial herpetofauna in the USVI consists of non-native species (Platenberg and Boulon 2006). Identifying the impacts of non-native reptiles and amphibians on native communities is measurable in some cases, as in the case of the brown treesnake (*Boiga irregularis*) in Guam (e.g., Rodda and Fritts 1992, Fritts and Rodda 1998,), the coqui frogs (*Eleutherodactylus* spp.) in Hawaii (e.g., Kraus et al. 1999, Beard and Pitt 2005), and the cane toad (*Bufo marinus*) in Australia (Phillips et al. 2003), among others. However, in many situations, identifying the impact of these invaders is difficult. Most of these species are easily overlooked and as such their date of establishment is generally unknown. They may have been quietly eating up all their competitors for some time before anyone noticed. These invaders are almost always accompanied by other compounding effects, such as habitat loss and fragmentation from development and agriculture, climate change, pollution, and other direct and indirect influences. Although there may be considerable anecdotal evidence attributing native species declines to a particular invader, actually proving this can be complicated.

The non-native herpetofauna in the USVI can be grouped into three categories according to their ostensible effect on native populations: high impact, presumed low impact, and unknown impact. Each of these categories are discussed in turn.

### High Impact Species

High impact species are those recently introduced, deliberately or accidentally, via human mechanisms from outside the region, and that share ecological niches or otherwise interfere with native species. Potential impacts include direct predation of native species, competition for limited resources (e.g., breeding sites, prey), and disease transmission. These species may also present other dangers to naïve species, through toxicity or other impacts. Examples of high impact species in the USVI include the Cuban treefrog, cane toad, and corn snake.

The Cuban treefrog has been present in the USVI since the 1970s (MacLean 1982) and is distributed across all three main islands (St. Thomas, St. John and St. Croix). This species is native to Cuba and nearby islands, and is thought to have invaded Florida, from where it has dispersed across the Caribbean. It arrived in the USVI as a stowaway in horticultural plants and construction materials, most likely from Florida. It exhibits remarkable abilities for colonization, hitch-hiking on a variety of transportation methods (Meshaka 1996b) and surviving under suboptimal and even harsh conditions (Townsend et al. 2000). It is omnivorous, and will eat anything it can shove into its mouth, including lizards with lengths exceeding that of the frog itself (personal observation). Diet studies of the Cuban treefrog have shown that it eats a wide variety of invertebrate species and will also take lizards, frogs and even conspecifics (Meshaka 1996a, Babbitt and Meshaka 2000, Owen 2005). This highly adaptable frog inhabits forests, woodlands, scrub, wetlands, and around human inhabitations, including cisterns and swimming pools (Meshaka 1996a). It requires freshwater for

reproduction, into which females can lay thousand of eggs per spawning occasion. Although there is little direct evidence of Cuban treefrog impacts on native herpetofauna in the USVI, there are considerable anecdotal assertions that where established, the native frogs and anole lizards have declined (E. Miles and several St. John residents, personal communication). The tadpoles likely interfere with the native white-lipped frog (*Leptodactylus albilabris*), the only native frog species to produce tadpoles (the others are direct developers), with which it shares ephemeral pools for breeding (personal observation).

The cane toad has been present since late 1800s or early 1900s, located primarily around agricultural areas on St. Thomas and St. Croix. This species was deliberately introduced to control agricultural pests, specifically beetle grub particularly damaging to crops (Van Volkenberg 1935). The cane toad is omnivorous, and will take any food available. This species requires a reliable fresh water source for reproduction, which restricts it in the USVI to man-made freshwater systems in disturbed habitats. The impact to native herpetofauna is unknown, although the tadpoles likely compete for limited freshwater resources with the white-lipped frog.

The corn snake was first reported in the territory in 2003 (Perry et al. 2003), although it has been present on St. Thomas since at least the mid 1990s, when an individual was collected and maintained in captivity (D. Griffin and R. Gomez, personal communication). This species was previously thought to be localized around the container port on the south-central side of St. Thomas to the west of the main commercial center of Charlotte Amalie, and until recently all sightings were restricted to this area. However, a road-killed gravid female was collected in a commercial area on the east side of the island in the summer of 2006, indicating some level of dispersal. The diet of this species in its native environment includes small mammals, frogs, birds, and lizards, and it is likely these snakes are taking advantage of the abundant anole lizards. They also pose a potential threat to the endangered USVI tree boa (*Epicrates monensis granti*) by direct predation and competition.

**Presumed Low Impact Species**

Presumed low impact species are those that do not share an obvious ecological niche or otherwise interfere with any native species. These species are often historically introduced, commonly as a food source, and tend to be herbivorous. They are generally novel, i.e., there is no ecologically similar native species. They present little competition for resources, although they may have indirect impacts through habitat alteration, disease introduction, or competition with non-similar species of other taxa. They may have other incidental and unknown impacts. Examples of these species in the USVI include the green iguana (*Iguana iguana*), red-footed tortoise (*Geochelone carbonaria*), house gecko, and freshwater turtles.

The origin of the green iguana in the USVI is unknown. It may be a deliberate 20$^{th}$ century introduction through the pet trade or well-intentioned attempts at conservation, or it may have been brought in by pre-Columbian Indians as a food source. It also may have dispersed naturally by rafting up the Antillean island chain. The green iguana is native to Central and South America, and is widely distributed across the West Indies (Schwartz and Henderson 1991). There is no documentation of the earliest specimens in the USVI. While there are iguanid fossil remains in prehistoric middens on St. Thomas (Pregill 1981, D. Brewer, personal communication), there is no evidence that there were any wild populations at that time. There are two species of the endemic West Indian iguanas on the Puerto Rico Bank (*Cyclura stejnegeri*, on Mona Island west of Puerto Rico, and *C. pinguis*, on Anegada at the eastern side of the British Virgin Islands; Schwartz and Henderson 1991); there is no evidence that these species ever occurred on the USVI, and, therefore, were not impacted by the arrival of the green iguana there. This large lizard is herbivorous and able to inhabit a wide variety of habitat types. They are very abundant on St. Thomas, particularly around resorts and open-air restaurants, where they feast on handouts and ornamental plants and lounge by the swimming pools, much to the dismay of the resort managers. They are often observed in association with the common ground lizard (*Ameiva exsul*), which may have discovered the benefits of following around a messy eater (personal observation). They were once considered locally endangered in the territory, and efforts were made to establish and augment populations on the main islands and even some of the satellite cays (Perry and Platenberg, In Press). They are still afforded special protection against killing and harm under local legislation (USVI Code Title 12).

The origin of the red-footed tortoise in the USVI is also a mystery. It is widely believed to have been introduced by pre-Colombian Indians as a food source. It inhabits dense forest and scrub habitat

and is seldom observed, and therefore its distribution in the territory is unknown. It is primarily herbivorous, and appears not to overlap ecologically with any similar native species.

The house gecko has been present in the USVI since 1700-1800s, likely having arrived via slave ships from eastern Africa. It is widely distributed across the Caribbean and into the southern US (Schwartz and Henderson 1991). It occurs on all the main islands and several satellite cays within the territory, although is most commonly found in association with human habitations; it feeds on insects attracted to lights at night. Although territorial, it can occur at high densities (Meshaka 2000). Its potential impact on native populations is through competition for invertebrate prey, although the nocturnal insects attracted to lights may not be readily available to diurnal native foragers.

Freshwater turtles, most commonly red-eared sliders (*Trachemys scripta elegans*), were not reported in the territory prior to the mid 1980s (MacLean 1982). They were most likely introduced into water bodies as unwanted pets. Long-time residents recall being able to buy them in pet shops (along with other exotic pets such as caiman). This species is highly aquatic, and as such have a very limited and human-influenced distribution. They inhabit nearly every freshwater body on St. Thomas and St. Croix, including agricultural ponds and ornamental pools at resort and golf courses. Almost all of these freshwater ponds are man-made because steep slopes and non-porous shallow soils preclude the natural formation of these systems. As such, there are no native species of freshwater turtles in the USVI. For the same reason, the native freshwater fish are amphidromous and restricted to stormwater drainages with occasional connection to the sea (Loftus 2003, Nemeth and Platenberg 2007), which are uninhabited by these turtles. The turtles may predate small waterfowl taking advantage of the rare freshwater resources, although they are observed to cohabit with little discernable effect (personal observation).

**Unknown Impact Species**

Species with an unknown potential impact are not native to the USVI, but are native within the region. They are species that have recently arrived, and their colonization may have resulted from a natural dispersive range extension, although in most cases the arrival of these species is human-mediated in a general trend for globalization. These species often have very similar counterparts present among the native fauna, and while their impacts are as yet unknown, they are likely to be significant. They are often "weedy" invaders, highly prolific and able to survive in a variety of habitats and under adverse conditions. Their impact on native populations has not yet been determined, although they are likely to impact native species through displacement of populations by direct competition for limited resources (prey and breeding sites), and may pose hybridization threats. There is also a highly likelihood of these individuals introducing diseases into a naïve population. Examples of these species in the USVI include the Puerto Rican coqui (*Eleutherodactylus coqui*), the Lesser Antillean whistling frog (*Eleutherodactylus johnstonei*), and the common ground lizard on St. Croix.

The Puerto Rican coqui is a recent immigrant to the USVI, presumed to have been introduced as recently as the 1990s in horticultural material from Puerto Rico. During its early years on St. Thomas, it appears to have been localized around the agricultural areas and garden centers at the highest elevation on Crown Mountain. In recent years, which have been moister than usual (USVI Division of Fish and Wildlife, unpublished data), the coqui has established populations in several locations on the island. Its habit of burying itself in potted plants enhances its dispersal (personal observation). These frogs can live at high densities from around 2000 to 20,000 individuals $ha^{-1}$ (Stewart and Woolbright 1996, Woolbright et al. 2006). It appears that they are displacing the native frogs (particularly, *Eleutherodactylus antillensis*) on St. Thomas, although this observation may be a seasonal artifact and requires further monitoring. The coqui has introduced the chytrid fungus (*Batrachochytrium dendrobatidis*), a significant cause of amphibian decline world-wide, to Hawaii (Beard and O'Neill 2005). The potential of this occurring in the USVI, which likely has temperatures considered too high for the survival of the fungus, requires investigation.

The Lesser Antillean whistling frog is native to many islands in the Lesser Antilles and is widely introduced across the West Indies and northern South America (Schwartz and Henderson 1991). Several specimens arrived in 2004 in cut flower shipments arriving from St. Vincent (unpublished data), but they appear not to have become established. The main concern is competition and hybridization with native frogs, as there is considerable overlap in habitat and diet (Ovaska 1991).

The common ground lizard is native to the Puerto Rican Shelf, where it is widespread,

abundant, and highly adaptive on main islands and satellite cays. This species has recently been introduced to St. Croix, where it previously never occurred. It is believed that the lizards were introduced through a shipment of car tires from either St. Thomas or Puerto Rico (W. Coles, personal communication), and they have since become established in one locality in the center of the island. The concern is that the lizard may make its way to the satellite islands (Ruth, Green, and Protestant cays) that harbor the only populations of the highly endangered St. Croix ground lizard (*Ameiva polops*), which has been extirpated from the main island and exist only in small, isolated populations (McNair 2003, McNair and Lombard 2004, McNair and Mackay 2005, Platenberg and Boulon 2006). It may also have an impact on the naïve St. Croix anole (*Anolis acutus*).

## ORIGIN AND MODE OF INTRODUCTION

The current demand for a "lush tropical" environment around resorts and upscale housing developments, instead of the scrubby dry subtropical vegetation native to the USVI, provides a continuous opportunity for species introductions. Produce and landscaping materials regularly arrive from Puerto Rico, Hispaniola, Florida, and other localities. There are two major container shippers in the USVI, as well as a plethora of other shipping companies servicing the US, West Indies, and beyond. One shipper visits 28 ports weekly across the Caribbean, as well as Canada and Florida (www.tropical.com). The residents of St. John assert that the coqui frog arrived on a single barge containing landscaping plants from Puerto Rico during the construction of an upscale resort on that island; a similar event occurred more recently (2006) when chorusing landscaping plants, also from Puerto Rico, were imported during the construction of a new marina on St. Thomas (personal observation). There are currently no provisions for inspecting cargo arriving from Puerto Rico or the US mainland, as these are considered "inter-state" transports and not subject to customs regulations. Only cursory inspections are carried out for goods arriving from other countries, although invasive vertebrate stowaways are not on the inspection radar. There are also no requirements for plant importers to inspect and monitor for non-native pests.

## RECOMMENDATIONS

It is acknowledged that many of the detrimental impacts to native wildlife by introduced species have already taken place, such as the extirpation of the Puerto Rican racer (*Alsophis portoricensis*) from the main island and the extinction of the St. Croix racer (*Alsophis sanctaecrucis*) by mongooses (Henderson and Powell 2001, Powell and Henderson 2005, Platenberg and Boulon 2006). It is also difficult to garner financial and staffing support for eradication of non-native species and prevention of new introductions, especially in the face of the more severe detrimental impacts of habitat loss and fragmentation, unregulated pesticide use, chemical contaminants, and other sources of environmental degradation. However, there are still native herpetofauna species that warrant protection from the threat posed by invasive species, and considering unknown impacts to other taxa a response is required. Some steps can be taken to reduce the incidence of introductions. Examples from the coqui invasion of Hawaii and the brown treesnake in Guam include increased inspection and showering landscaping plants with hot water (www.ctahr.hawaii.edu/coqui/shower.asp). Regulations can be imposed to require landscapers and importers to inspect and monitor shipments. Stricter permitting can limit the importation of exotic pets that inevitably end up free-roaming. A rapid response team could be established. However, the single most effective mechanism would be to educate the public and merchants as to the consequences of the non-native species and to enlist their vigilance and stewardship.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

BABBITT, K. J., AND W. E. MESHAKA, JR. 2000. Benefits of eating conspecifics: effects of background diet on survival and metamorphosis in the Cuban treefrog (*Osteopilus septentrionalis*). Copeia 2000:469-474.

BEARD, K. H., AND E. M. O'NEILL. 2005. Infection of an invasive frog *Eleutherodactylus coqui* by the chytrid fungus *Batrachochytrium dendrobatidis* in Hawaii. Biological Conservation 126:591-595.

BEARD, K. H., AND W. C. PITT. 2005. Potential consequences of the coqui frog invasion in Hawaii. Diversity and Distributions 11:427-433.

BREWER, D. M. 2005. Cultural history of the Virgin Islands: a short primer. Pages 19-23 *in* T. Thomas and B. Devine, editors. Island peak to coral reef: a field guide to the plant and marine communities of the Virgin Islands. University of the Virgin Islands Press, St. Thomas.

FRITTS, T. H., AND G. H. RODDA. 1998. The role of introduced species in the degradation of island ecosystems: a case history of Guam. Annual Review of Ecology and Systematics 29:113-140.

HENDERSON, R. W., AND R. POWELL. 2001. Responses by the West Indian herpetofauna to human-influenced resources. Caribbean Journal of Science 37:41-54.

KRAUS, F., E. W. CAMPBELL, A. ALLISON, AND T. PRATT. 1999. *Eleutherodactylus* frog introductions to Hawaii. Herpetological Review 30:21-25.

LOFTUS, W. F. 2003. Inventory of fishes in inland fresh- and brackish-water habitats of Virgin Islands National Park. Final report. U.S. National Park Service Inventory and Monitoring Program, Southeast Region, Atlanta, Georgia, USA.

MACLEAN, W. P. 1982. Reptiles and amphibians of the Virgin Islands. Macmillan Education Ltd., London.

MCNAIR, D. B. 2003. Population estimate, habitat associations, and conservation of the St. Croix ground lizard *Ameiva polops* at Protestant Cay, United States Virgin Islands. Caribbean Journal of Science 39:94-99.

MCNAIR, D. B., AND C. D. LOMBARD. 2004. Population estimates, habitat associations, and management of *Ameiva polops* (Cope) at Green Cay, United States Virgin Islands. Caribbean Journal of Science 40:353-361.

MCNAIR, D. B., AND A. MACKAY. 2005. Population estimates and management of *Ameiva polops* (Cope) at Ruth Island, United States Virgin Islands. Caribbean Journal of Science 41:352-357.

MESHAKA, W. E., JR. 1996a. Diet and colonization of buildings by the Cuban treefrog, *Osteopilus septentrionalis* (Anura: Hylidae). Caribbean Journal of Science 32:59-63.

MESHAKA, W. E., JR. 1996b. Vagility and the Florida distribution of the Cuban treefrog (*Osteopilus septentrionalis*). Herpetological Review 27:37-39.

MESHAKA, W. E., JR. 2000. Colonization dynamics of two exotic geckos (*Hemidactylus garnotii* and *H. mabouia*) in Everglades National Park. Journal of Herpetology 34:163-168.

NEMETH, D., AND R. PLATENBERG. 2007. Diversity of freshwater fish and crustaceans of St. Thomas watersheds and its relationship to water quality as affected by residential and commercial development. Water Resources Research Institute Project 2006VI73B, University of the Virgin Islands, St. Thomas.

OVASKA, K. 1991. Diet of the frog *Eleutherodactylus johnstonei* (Leptodactylidae) in Barbados, West Indies. Journal of Herpetology 25:486-488.

OWEN, J. 2005. The Cuban treefrog (*Osteopilus septentrionalis*): diet, reproduction, and distribution of an invasive species in the British Virgin Islands. MS thesis. Texas Tech University, Lubbock, Texas, USA.

PERRY, G., J. PIERCE, D. GRIFFIN, G. VAN BUURT, AND J. LAZELL. 2003. *Elaphe guttata guttata*. Herpetological Review 34:264-265.

PERRY, G., AND R. PLATENBERG. In press. Recent additions to the herpetofauna of Little St. James, U.S. Virgin Islands. Applied Herpetology.

PHILLIPS, B. L., G. P. BROWN, AND R. SHINE. 2003. Assessing the potential impact of cane toads on Australian snakes. Conservation Biology 17:1738-1747.

PLATENBERG, R. J., AND R. H. BOULON, JR. 2006. Conservation status of reptiles and amphibians in the U.S. Virgin Islands. Applied Herpetology 3:215-235.

PLATENBERG, R. J., F. E. HAYES, D. B. MCNAIR, AND J. J. PIERCE. 2005. A comprehensive wildlife conservation strategy for the U.S. Virgin Islands. Division of Fish and Wildlife, Department of Planning and Natural Resources.

POWELL, R., AND R. W. HENDERSON. 2005. Conservation status of Lesser Antillean reptiles. Iguana 12:2-17.

PREGILL, G. 1981. Late Pleistocene herpetofaunas from Puerto Rico. Miscellaneous Publications No.71. University of Kansas Publication, Museum of Natural History, Lawrence, Kansas, USA.

RODDA, G. H., AND T. H. FRITTS. 1992. The impact of the introduction of the colubrid snake *Boiga irregularis* on Guam's lizards. Journal of Herpetology 26:166-174.

ROGOZIŃSKI, J. 1999. A brief history of the Caribbean: from the Arawak and the Carib to the present. Plume, New York, USA.

SCHWARTZ, A., AND R. W. HENDERSON. 1991. Amphibians and reptiles of the West Indies: descriptions, distributions, and natural history. University of Florida Press, Gainesville, Florida, USA.

STEWART, M. M., AND L. L. WOOLBRIGHT. 1996. Amphibians. Pages 273-320 *in* D. P. Reagan and R. B. Waide, editors. The food web of a tropical rain forest. University of Chicago Press, Chicago, Illinois, USA.

TOWNSEND, J. H., J. M. EATON, R. POWELL, J. S. PARMERLEE, JR., AND R. W. HENDERSON. 2000. Cuban treefrogs (*Osteopilus septentrionalis*) in Anguilla, Lesser Antilles. Caribbean Journal of Science 36:326-328.

USDA-NRCS. 1998. Soil survey of the United States Virgin Islands, United States Department of Agriculture, Natural Resources Conservation Service, in cooperation with the Virgin Islands

Department of Planning and Natural Resources; the Virgin Islands Cooperative Extension Service; and the United States Department of Interior, National Park Service.

VAN VOLKENBERG, H. L. 1935. Biological control of an insect pest by a toad. Science 82:278-279.

WOOLBRIGHT, L. L., A. H. HARA, C. M. JACOBSEN, W. J. MAUTZ, AND F. L. BENEVIDES, JR. 2006. Population densities of the coqui, *Eleutherodactylus coqui* (Anura: Leptodactylidae) in newly invaded Hawaii and in native Puerto Rico. Journal of Herpetology 40:122-126.

# THE BROWN TREESNAKE RAPID RESPONSE TEAM

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***Abstract:*** In the 1940s the brown treesnake (*Boiga irregularis*) was accidentally transported to Guam and became established. Brown treesnakes have caused and continue to cause major problems for the ecology, economy, and quality of life on Guam. As Guam's snake densities increased in the late 1970s, extralimital encounters began to be reported on islands with transportation links to Guam. In 1993, a major effort was initiated to reduce the potential for brown treesnakes to accidentally enter Guam's transportation system. In 2002, a multi-agency Rapid Response Team (RRT) was established to assist in detection and capture of brown treesnakes on recipient islands after being accidentally transported from Guam. Since its creation, the RRT has conducted 2-5 training courses annually on Guam and 16 off-Guam field operations. As of 2006, the RRT consisted of 66 members located throughout the Pacific region and the United States mainland. The RRT has incorporated research results from affiliated agencies, such as means to improve snake detectability at low densities, effectiveness of control tools in rodent-rich environments, and predicting movements of snakes accidentally translocated. The RRT continues to work with regional island groups, improving communication networks, elevating public awareness, and developing response capabilities.

***Key Words:*** *Boiga irregularis*, brown treesnake, extralimital populations, Guam, Hawaii, invasive species, management, Micronesia, rapid response.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Invasive species are a global concern, with numerous species causing significant damage or potential damage to countless systems (Campbell 1991, Wilcove et al. 1998, Pimentel et al. 2005). As global trade continues to increase, so does the potential for accidental transportation and colonization by invasive species (Jenkins 1996, Christy et al. 2007). Once established, invasives are costly to economies, damaging to natural resources, and often uncontrollable or exceedingly expensive to control (Pimentel et al. 2005, Burnett et al. 2006). Therefore, preventing the spread of invasive species is preferable to control or eradication after establishment (Simberloff et al. 2005). In cases of a suspected introduction, reacting quickly and effectively to minimize or eliminate new populations is preferable to taking no, or ineffective, action. An economic analysis of brown treesnake (*Boiga irregularis*, Colubridae, BTS) management on Oahu indicated that much greater effort should be expended on eliminating or containing incipient populations on recipient islands (Burnett 2007).

The brown treesnake is a slender snake native to the area extending east of Wallace's Line to the Solomon Islands as well as the humid northern and eastern Australian coasts (Rodda et al. 1999b). BTS are nocturnal and oviparous. In their native range, BTS appear to be relatively common (Rodda et al. 1999b), and are not known to cause any major conflicts with the environment or humans as they are presumably in equilibrium within the natural system. The central component of this equilibrium is thought to be low prey availability (Rodda et al. 1999c). BTS are good climbers, seek refuge during daytime heat and light, and search at night for prey items such as birds, lizards, and rodents (Rodda et al. 1999b).

Brown treesnakes were accidentally transported to Guam (Savidge 1987, Fritts 1988, Rodda and Fritts 1992), most likely arriving on Guam as accidental stowaways in the late 1940s, when salvaged military equipment was shipped from the Admiralty Islands (Fritts and Rodda 1998). The population on Guam is the only confirmed breeding population outside of its native range. In the decades since its arrival, the BTS has severely impacted the economy and ecology of Guam, as well as having direct impacts on human health (Savidge 1987, Fritts et al. 1990, Fritts and McCoid 1991, Perry and Morton 1999, Fritts 2002).

Lacking geographical or ecological barriers, the BTS spread throughout the island, by the mid-1980s had reached unprecedented densities of 50-100 snakes per ha in some localities (Rodda et al. 1999b, Fritts 2002). Population densities have since dropped, but overall densities remain substantial (Rodda and Fritts 1992, Rodda et al. 1999b). In 1990, the BTS was declared an Injurious Wildlife species (Federal Register 55(80):17439-17441, McCoid et al. 1994), prohibiting its intentional importation into the United States (US) for most commercial purposes. However, intentional importation may be a relatively minor pathway for BTS dispersal as it makes a poor pet, being drab, secretive and prone to biting.

Guam is a hub for commercial and military shipments in the tropical western Pacific. High levels of transportation with regional and external locations greatly increases the threat of BTS being transported from Guam to new locations. In 1993, the US Department of Agriculture, Wildlife Services (WS), in conjunction with other agencies, including Guam Department of Agriculture, Division of Aquatic and Wildlife Resources (GDAWR), began a concerted interdiction program to reduce the potential for snakes to accidentally enter Guam's transportation system (Vice et al. 2005). WS has taken the primary role in this effort through trapping, oral toxicants, fence line searches and the use of BTS detection dog teams. The protection of port facilities and the examination of outbound cargo and planes require an extensive but highly cost-effective effort. This effort has minimized the probability of snakes being transported from Guam to new locations. While these efforts have dramatically reduced the likelihood of such an event (Vice et al. 1999, Engeman and Vice 2002, Vice and Vice 2004), no inspection program can be 100% effective, as all control tools experience occasional failures and some shippers actively evade the voluntary inspection program.

To further reduce the probability of BTS becoming established on islands receiving goods from Guam, the highest risk islands (Saipan, Oahu, and Tinian) have established inbound interdiction protocols, emphasizing dog-aided inspections of cargo coming from Guam. Sentinel traps have been placed within port areas in the Northern Mariana Islands to assist detection and capture of arriving BTS. However, snakes may be unmotivated or unable to exit cargo at the port of entry, only to escape at the cargo's final destination or other interior localities. It is not economically feasible to maintain permanent interdiction measures at all potential cargo destinations on a given island. This limitation, interdiction failures, and post-interdiction dispersal are presumably responsible for the 123 credible BTS encounters recorded to date at recipient locations.

## EXTRALIMITAL SIGHTINGS

Apart from encounters on Guam and one on Wake Island (1949), reports of BTS outside of their native range began in the late 1970s and continue to the present (Figure 1). Numerous recipient island BTS encounters are associated with cargo from Guam, and it is generally accepted that BTS encounters in non-native range locations are likely snakes transported from Guam. The probability of snake transport from Guam is much higher than from its native range, due to Guam's high snake densities and the high frequency of ship and plane traffic departing Guam. In this paper, sightings of BTS outside of Guam and the snake's native range will be termed extralimital or recipient-island encounters.

The increase in reported extralimital encounters from 1978 to 1992 (Figure 1) was presumably due to a combination of high snake densities on Guam, increased cargo traffic from Guam, and improved reporting of encounters in recipient locations. Since the mid-1980s, the public was encouraged to report extralimital BTS sightings to authorities, with greatly improved public outreach efforts since 2000 (Hawley 2007, Martin 2007). As a result, the percentage of sightings reported to officials has probably increased, with actual sightings more numerous than suggested by Figure 1. The under-reporting bias appears most severe in earlier years, when awareness levels were substantially lower.

In the most recent 5-year time period (2003-2007) there were fewer extralimital BTS encounters than in comparable periods of the preceding decade. It is unknown whether this decrease in encounter reports represents a long-term trend or merely a fluctuation. We believe that a reduction of snake transport events has occurred, given the greatly-increased BTS interdiction effort on Guam after 1994. Some encounters reported after 1994 may reflect snakes (or their progeny) transported prior to 1994, but discovered later. All recent sightings, however, are not due to pre-1994 transport in cargo.

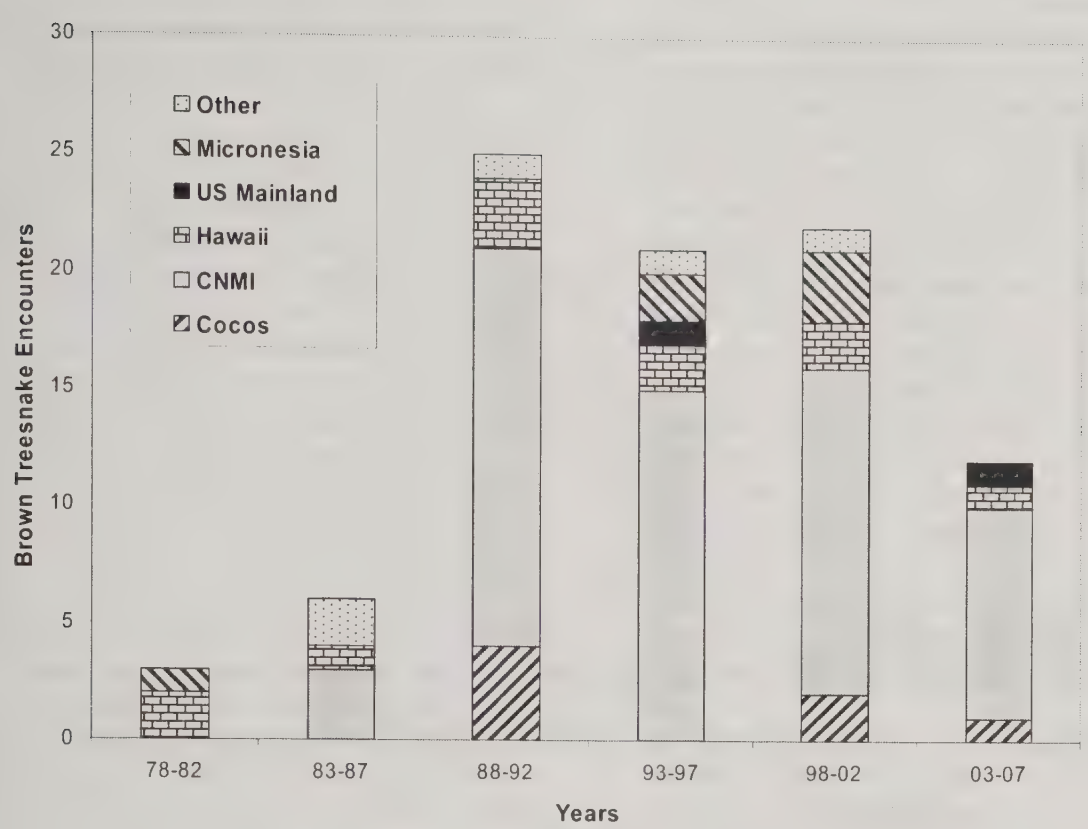



**Figure 1.** Brown treesnake encounter reports. Additional encounters not shown include Wake Island (1949), 1 each for Alaska and Taiwan (dates unknown), and 31 reports from the Northern Mariana Islands with unknown dates.

While encounters have occurred in numerous locations (Figure 2), they tend to be concentrated on a few high-risk islands such as Saipan, Tinian,and Oahu (Fritts 1987, Fritts 1988, McCoid and Stinson 1991, McCoid et al. 1994, Fritts et al. 1999). Of the 76 reports from Saipan, 13 are captures and 63 are credible sightings (many additional reports are insufficiently documented to warrant characterization as "credible"). The large number of encounters on Saipan has led to speculation that a breeding population may already be established (Colvin et al. 2005).

Aside from the high frequency of BTS encounters on Saipan, the island is of special concern because it receives much cargo from Guam, and the other Mariana Islands receive most of their cargo from Saipan. Thus, infestation of Saipan is both possible and consequential. Saipan is the largest snake-free island in the Marianas, and it is the primary refuge for many bird species (or sister taxa) that have been extirpated on Guam. Saipan and surrounding islands are the sole remaining refuge for the Marianas rufous fantail (*Rhipidura rufifrons saipanensis*), Mariana fruit-dove (*Ptilinopus roseicapilla*), Saipan bridled white-eye (*Zosterops saypani*), Saipan nightingale reed-warbler (*Acrocephalus luscinia nijoi*), Tinian monarch (*Monarcha takatsukasae*), and the golden white-eye (*Cleptornis marchei*).

The island with the second-largest number of encounters is Oahu (Figure 2). It shares with Saipan the attribute that should the island become infested, the large amount of local inter-island cargo traffic would render it very problematic to keep an infestation from spreading to adjacent islands (in Oahu's case, the remainder of the Hawaiian Islands).

## POST-DISPERSAL CONTROL OF BROWN TREESNAKES ON RECIPIENT ISLANDS

As previously mentioned, permanent interdiction facilities at each possible cargo destination from Guam are impractical. In many states, the response to such a situation is often to abandon further control efforts as being too costly.

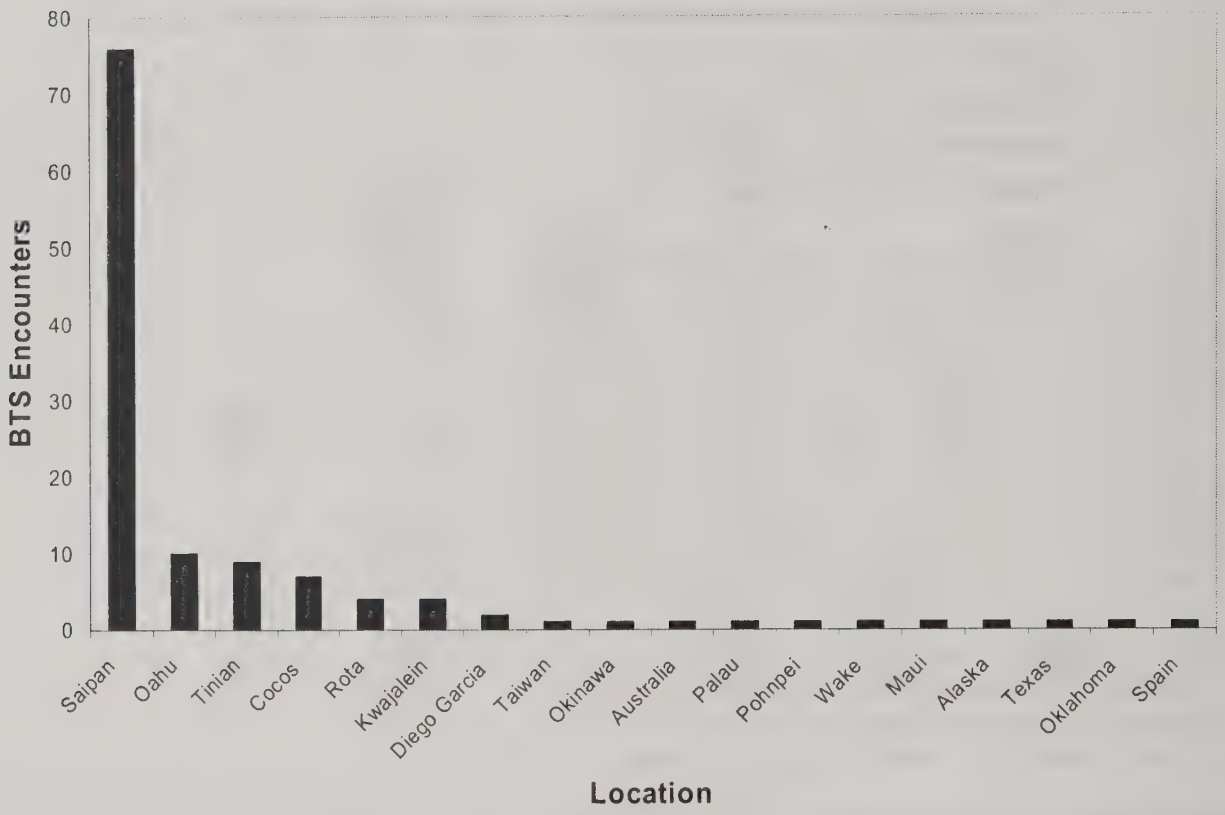



**Figure 2.** Encounter reports by location for brown treesnakes thought to have originated from Guam (excluding the single Wake Island report).

However, control tools for BTS are better developed than for almost any other reptile, and the ecological and economic costs of inaction are great. From 1990 to 2002, recipient island responses ranged from inaction to temporarily deploying a small number of snake traps in the vicinity of encounters, coupled with a few night-time visual searches in the area conducted by local wildlife personnel.

Research activities on Guam gradually revealed a number of key weaknesses with this approach: (1) dispersing snakes are mostly small, and the snake traps currently available fail to catch small snakes, (2) snake traps baited with live mice are not effective in environments with high rodent densities which we expect to find on most recipient islands, and (3) visual searcher abilities are highly variable.

Vice and Vice (2004) showed that BTS removed from the transportation network were relatively small: 83% were smaller than 900 mm SVL (snout-vent length). Rodda et al. (2007) demonstrated that conventional BTS traps are largely ineffective for snakes smaller than 900 mm SVL with over 25% of BTS in large-scale trapping less than 800 mm SVL (WS, unpublished data). Thus, we would not expect traps used on recipient islands to be effective, and in fact, extralimital BTS trapping efforts to date have captured no snakes, nor have visual searches observed any snakes.

Rodda et al. (2001) looked at the relationship between rodent abundance and snake trap capture success, and found a seven-fold decrease in trap effectiveness in areas of high rodent abundance on Guam. However, many recipient islands are suspected of having even higher rodent abundances, implying that trap success might be further reduced on recipient islands. Wiewel et al. (2006) demonstrated that many recipient island sites had higher rat (*Rattus* spp.) densities than eight comparable Guam sites. Gragg et al. (In Press) demonstrated experimentally that rodent reduction on Guam elicited a rapid (~ one week delay) 38-65% increase in trap capture success following rodenticide application. Thus, traps

would have relatively poor prospects of success for snakes of any size on high rodent density islands such as Saipan, unless rodent contral is integrated into response protocols.

Researchers have long been aware that some searchers are more effective than others when searching for snakes in the forest. Brown treesnakes are vine-like in appearance and their behavior does not facilitate visual detection (Rodda and Fritts 1992). These authors found a 10-fold range in effectiveness of trained searchers, and a 24-fold difference in a later study (1995-99, unpublished data) based on at least 25 hours of search time. We assume that inexperienced searchers are less effective than trained searchers, but it is difficult to get an adequate sample of inexperienced searcher effectiveness. As searchers develop experience over the time needed for quantification they become experienced before the "inexperienced" sample is completed. The specific challenge for searchers on recipient islands (except Palau) is that there are no native snakes on which to practice searching. Locally-trained searchers on recipient islands, therefore, may not be as effective as they would be if trained on Guam.

If traps and locally-trained searchers are not effective responses to dispersed snakes on recipient islands, what is a wildlife manager to do? Validation of searchers trained on Guam provides a partial answer. Rodda et al. (2007) demonstrated that searchers trained on Guam were able to find both large and small snakes, and all snakes in a geographically closed 5-ha population could be detected given sufficient effort (59 searches, 826 search-hr). All resident small snakes were found in the first 12 searches. Snake trapping in areas where rodent densities have been suppressed by rodenticide is another tool that can be used on recipient islands. Investigations into the effectiveness of specially-trained dogs for detection of snakes at low densities are ongoing.

## US GEOLOGICAL SURVEY BROWN TREESNAKE RAPID RESPONSE TEAM

The US Geological Survey's (USGS) Brown Treesnake Rapid Response Team (RRT) was established in 2002 at the request of the US Department of Interior's Office of Insular Affairs. The RRT was established to: (1) train searchers on Guam, (2) provide experienced searchers for extralimital searches, (3) assist in communicating new developments in BTS science to recipient islands, and (4) provide guidance in the conduct of responses to BTS sightings. Placement of these tasks within USGS, a research agency, was justified by the absence of trained visual searchers in other agencies. The RRT is administered by the USGS RRT Coordinator (RRTC), and membership includes US federal, state, and territorial agency personnel, as well as personnel from several foreign governments. Active members are those that have been trained and undergone subsequent refresher training as needed. Current team members include staff from the following agencies: USGS, WS, GDAWR, US Fish and Wildlife Service, Northern Mariana Island's Division of Fish and Wildlife, Federated States of Micronesia's Division of Agriculture, Marshall Islands' Ministry of Resources and Development, Palau Division of Agriculture, Hawaii Department of Agriculture, Hawaii Department of Land and Natural Resources, Coordinating Group on Alien Pest Species, Maui Invasive Species Committee, and Colorado State University.

Team membership and training (e.g., the number of islands with trained team members) continue to increase (Figure 3), with 66 members from 15 locations at the end of 2006 (Figure 4). These personnel are made available by their agencies for response actions. The majority of team members are stationed within the Marianas (12) and Hawaiian (25) archipelagos. We expect that an additional 10 team members will be trained in 2007, mainly from Hawaii and the Marshall Islands.

Since its inception in 2002, the brown treesnake RRT has provided Guam-based searchers for 16 BTS responses (Table 1). Most of these responses have been to the Northern Mariana Islands (14), specifically Saipan (8). Responses have also occurred on Maui and Pohnpei. The most active year was 2003, with 7 responses occurring in the Northern Mariana Islands.

Proactive coordination among the various recipient islands and a consistent interview process are necessary for optimal response to a snake sighting. Prior to the development of a coordinated RRT effort, each island had its own approach to responding to a sighting. Spurious sightings, incomplete interviews of the person reporting the sighting, and ad hoc responses made it difficult to assess the credibility of sightings and evaluate the effectiveness of the response. The first step towards addressing this inter-island inconsistency was to develop a standard snake sighting interview protocol (Appendix A), which uses props and illustrations to avoid leading the witness.

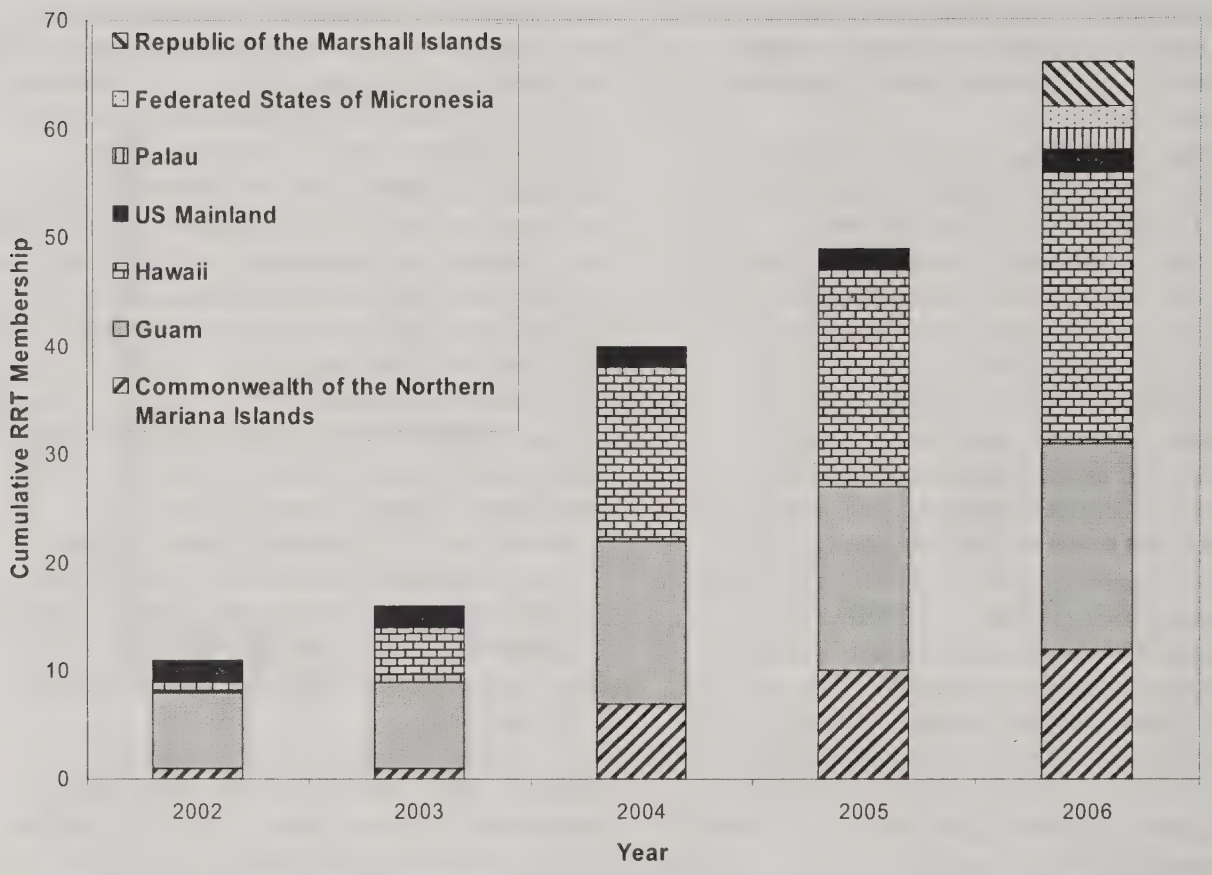



**Figure 3.** Rapid Response Team cumulative membership from its inception in 2002-2006.

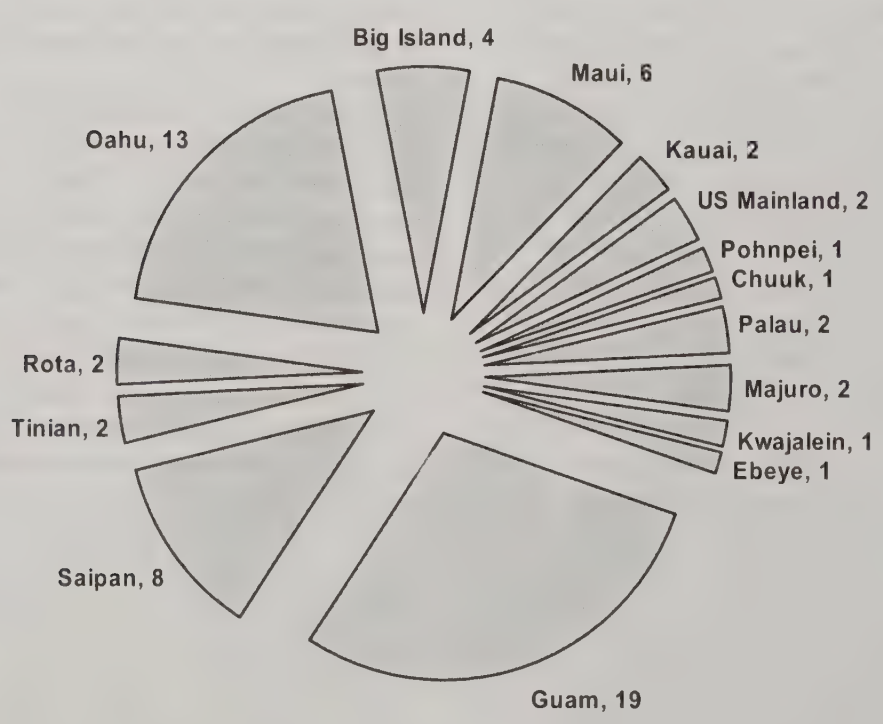



**Figure 4.** Rapid Response Team membership by island for 2006.

**Table 1.** US Geological Survey Rapid Response Team response actions to date.

| Date | Location | Agencies Involved | Reason for Response |
|---|---|---|---|
| 29-May-02 | Rota | CNMI DFW, USGS | BTS sighting report |
| 18-Dec-02 | Rota | CNMI DFW, USGS | Typhoon damage/ transportation concern |
| 24-Feb-03 | Saipan | CNMI DFW, USGS | Search for incipient BTS population |
| 20-Apr-03 | Saipan | CNMI DFW, USGS, HIDOA, HIDLNR | BTS sighting report |
| 31-Jul-03 | Saipan | CNMI DFW, USGS | BTS sighting report |
| 10-Aug-03 | Tinian | CNMI DFW, USGS | BTS sighting report |
| 20-Sep-03 | Tinian | CNMI DFW, USGS | BTS sighting report |
| 8-Nov-03 | Tinian | CNMI DFW, USGS | BTS sighting report |
| 9-Nov-03 | Tinian | CNMI DFW, USGS | BTS sighting report |
| 10-Aug-04 | Maui | HIDOA, HIDLNR, MISC, HIDOH, USGS, CNMI DFW | BTS sighting report |
| 22-Sep-04 | Saipan | CNMI DFW, USGS | BTS sighting report |
| 28-Sep-04 | Saipan | CNMI DFW, USGS | BTS sighting report |
| 20-Mar-05 | Saipan | CNMI DFW, USGS | BTS sighting report |
| 21-May-06 | Pohnpei | FSM Agr, USGS | BTS sighting report (later determined to be a blind snake) |
| 3-Jan-07 | Saipan | CNMI DFW, USGS | BTS sighting report |
| 20-Feb-07 | Saipan | CNMI DFW, USGS, GDAWR, USFWS | Search for incipient BTS population |

CNMI DFW = Commonwealth of the Northern Mariana Island Division of Fish and Wildlife, HIDOA = Hawaii Department of Agriculture, HIDLNR = Hawaii Department of Land and Natural Resources, MISC = Maui Invasive Species Committee, HIDOH = Hawaii Department of Health, FSM Agr = Federated Sates of Micronesia Division of Agriculture, GDAWR = Guam Department of Agriculture, USFWS = United States Fish and Wildlife Service.

Witnesses may be intimidated by questions from government officials, and may provide the answer that they think the official wants. When possible, the interview is administered by a local person using the witness's first language. Yes/no questions are avoided (as "yes" answers tend to be given disproportionately) and distances or sizes are given with reference to physical models rather than measured units. For example, the witness may be asked to match the thickness of the snake to the closest match from a series of ropes of different thicknesses; they are not asked for measurements. In the western Pacific, the taxonomic target of each interview is the BTS, so a variety of BTS photos are used as interview props. However, in Hawaii the taxonomic range of likely snake species is broad, calling for a greater diversity of photographic props. The interview questions are invariant across the Pacific. Each witness is given the opportunity to request confidentiality for individual answers. For example, the exact sighting location can be withheld from the public, as might the witness's identifying information (name, phone, address, etc.).

The USGS is in the process of posting the full snake sighting database (with appropriate limitations for confidentiality) on a public internet site. The site will have mapping capability such that anyone can point to a spot on a map and see locations, dates, and data for nearby snake sightings. This is useful for discerning whether a cluster of sightings has occurred in any particular venue.

A cluster of sightings is of special interest in that it suggests that an incipient snake population has a higher probability of existing in that area. Single credible sightings generally evoke a search effort involving approximately twelve searchers for two weeks. This is the most rapid action the RRT implements. Less credible sightings rarely involve deployments of searchers from Guam, but may justify a smaller search effort by local searchers (credibility is rarely clear-cut, and one can rarely be confident that a snake population does not exist if

one has not looked). A cluster of sightings evokes a much stronger response, but with less urgency, as the cluster is not based on any single stimulus. Response to a cluster is critical, as it reflects growing evidence that a snake population may be present. To date, there has been only one putative cluster identified (west of Saipan airport, searched intensively in February 2007). Response to the airport cluster comprised participation by 49 visual searchers over 21 nights, as well as hundreds of traps and four dog teams. Thus, a response is variable and is calibrated to address the perceived snake colonization risk.

A possible misinterpretation of the RRT concerns the objective of response actions. Responses are usually timed to follow immediately after a snake sighting is reported, but that misleads some into assuming that the sole purpose of a response is to locate the reported snake. While capture of the snake that was sighted would be desirable (though it has yet to occur), the more important objective is to determine whether a snake population exists in the area where the sighting or cluster was discovered. Single snake sightings constitute an indication that a population may have developed, but it is the population rather than the individual that is of most concern.

Estimating the geographic extent and population size of an incipient population (both geographic and numeric) is crucial. Current technology suffices to eradicate BTS from small areas at reasonable cost (Rodda et al. 1998, 1999a), but it is unclear whether BTS can be eradicated from large areas, and such an accomplishment may be prohibitively expensive with currently available technology. Thus, identifying a population before it has grown beyond a small area is critical.

Fortunately, a population is much easier to detect than is a single individual. Capture probability results from marked populations on Guam indicate that a single snake might not be found during the brief period of a response. Unlike most BTS research sites, many sighting locations are extremely difficult to inspect (e.g., debris piles, dense vine tangles, habitations and structures), and a snake may leave the sighting vicinity either by chance or in response to some stimuli. The RRT has only a qualified expectation that the reported snake will be recovered during the brief period of a response. However, rigorous estimates of capture probability on Guam indicate a very strong likelihood that a snake will be seen if there is a population of snakes in the area searched. The exact probability depends on the number of snakes in the population, the difficulty of searching a specific site, the amount of effort applied to the search, and the level of snake activity as a function of resource availability. Under conditions tested on Guam, even a single snake that remains in the search area would likely be detected by the effort associated with an ordinary RRT response.

Our main concern is the establishment of an incipient population, but the RRT is also interested in capturing any reported snake. The best odds of encountering a particular snake are achieved by responding rapidly to sighting reports. In addition to increasing the potential for encountering a reported snake, rapid reaction to a public sighting highlights the crisis aspect of BTS sightings and increases public appreciation and understanding of the threat posed by these snakes.

Public awareness and reporting is critical to the functionality of the response team. While a RRT response may put dozens of eyes in the field at the right time and place, the public constitutes thousands of observers that spend their lives in that area. The value of the public in spotting and reporting snakes cannot be overstated. Without sighting reports, wildlife managers would likely be unaware of incipient populations until they have grown too large to eradicate. The probability of finding and capturing a single snake is greatly reduced with the passing of time, as the potential search area grows exponentially. In locations where educational outreach is minimal or non-existent, sightings are often unreported. If they are reported, reports are frequently submitted days or weeks after the sighting. Outreach to the public is critical in high risk areas but is currently lacking or minimal on islands other than Hawaii and the Northern Marianas The RRTC is currently working with several island groups to increase their outreach capacities.

The RRTC also has a role to play in assisting responsible local authorities with developing institutional mechanisms for preventing and addressing the BTS invasion threat. For example, many remote island officials may not be fully aware of what BTS have done on Guam, and may not have a 24/7 phone number for the public to report sightings. They may not have a set protocol for evaluating sighting reports and developing a response. There may also be jurisdictional uncertainty about whether a forestry, wildlife, agriculture, or quarantine agency has lead responsibility. While the RRTC does not purport to have the definitive answer to these sorts of questions, he/she has witnessed similar discussions

on many islands and can convey which solutions have been tried elsewhere, and how successful they were. Network building and preparation of first responders is a key step and the RRTC assists with training when requested by local authorities.

A core responsibility of the RRTC is the training on Guam of recipient island personnel. Generally, initial training for response team members entails an intensive 18-day program. These trainings are held 1-3 times a year depending on the needs of cooperators. Training focuses on development of appropriate BTS search images and acquiring a basic skill set which increases each team member's ability to respond appropriately to a sighting. Suitable methods for the capture and containment of venomous and non-venomous snakes, proper use of resources, interviewing a sighting reporter, navigation in remote locations, setting up preliminary search areas, search strategies, and response documentation are some of the topics covered during a typical training course.

Refresher training courses are also held yearly. These courses are designed for previously trained response members (it is strongly urged that all team members receive updated training every two years) and focus on maintaining an appropriate BTS search image. Refresher courses also cover topics as requested by cooperators and update team members on the most current theories and resources used during a response.

## BROWN TREESNAKE RESPONSE MANAGEMENT

The response to a BTS sighting is up to local authorities. Communication with the RRTC is strongly encouraged, but strictly voluntary. Local authorities may request consultation, the direct involvement of off-island searchers, or full participation by the RRT. The availability of RRT resources is dependent on conflicting demands and funding availability, but we endeavor to provide as much assistance as funding and local interest warrants.

There is no one correct way to respond to a sighting report; however, there are certain commonalities. If an initial report is suggestive of a BTS sighting, and trained or competent searchers are locally available, we suggest that such searchers travel immediately to the sighting location to begin visual searches. Immediate response heightens the prospects for capturing the reported snake, and in at least one case on Oahu the snake (not a brown treesnake) was retrieved by timely arrival of searchers.

A second priority is to comprehensively evaluate the probability that the sighting constitutes evidence of a BTS population or individual. This is accomplished through a detailed interview of the reporting person, preferably documented by a trained team member at the sighting location, although other formats such as telephone interviews are also acceptable (see appendix A for the standard interview form). The interview should be conducted as soon as possible, preferably within a few hours of the sighting. Rapid interviews benefit from fresher memories, additional context, and may preempt inappropriate responses based on premature judgments (e.g., prematurely flying in off-island searchers).

The interviewer will be asked for his/her evaluation of the probability that the sighted object was a BTS, but in a situation in which the local interviewer has relatively little personal experience with BTS, the obtained information should also be vetted by experts familiar with BTS appearance and behavior. Additional assessments can also be obtained by email from experts identified by the RRTC. In many cases, interview results are relatively unequivocal, indicating that the sighted organism was a blind snake (*Ramphotyphlops braminus*) or monitor lizard (*Varanus indicus*). In many cases deemed "credible", the evidence supports the likelihood that the sighting object was a BTS.

The brown treesnake RRT is funded primarily for response to BTS sightings, but if credible information regarding a non-BTS sighting is received and the host agency requests RRT assistance, it will be provided dependent on expertise that can be offered and availability of resources. When the species of snake is questionable, it is best to cautiously treat the report as a BTS sighting. However, for safety, when capturing an unknown species of snake, we assume it is venomous and handle it as such. If off-island RRT searchers are requested, warranted, and available, a response team will depart from Guam for the sighting location on the next available flight (usually within a day of the assistance request). In general, 3-6 trained members from Guam, including the RRTC, are available to travel to the sighting location. The RRT may also supply resources such as traps and dog teams depending on requests and availability. It is expected that the local host agency will supply additional field personnel. If warranted, more trained team

members may be requested from either Guam or other locations to assist with a response.

## INCIDENT COMMAND STRUCTURE

Opportunities for chaos abound during a typical response action making pre-planning critical in order to avoid potential pitfalls. Some of the challenges during a response include: (1) the need for rapid action which generally involves personnel from multiple agencies and locations, (2) addressing landowner/property rights as needed in the sighting location, (3) coordinating field crews and response tools including visual searchers who are in the field after sunset, (4) participation from both print and video media, (5) tracking what has and has not been searched/cleared/prepped, and (6) mapping the response area and associated efforts.

To minimize frustration and overtime costs, maximize volunteer enthusiasm, assure searcher safety, and optimize educational outreach through the inevitable media attention, it is imperative that a response be well organized. In general, the solution to all of these problems is to develop an appropriate chain of command and record-keeping procedures before the incident, and convince the participants of the importance of disciplined adherence to the plan when incidents arise. Lines of responsibility and communication need to be clearly drawn and rigorously adhered to. Media requests should be channeled through a single local authority (who may choose to involve others).

The standard method for organizing events of this nature is called "incident command structure," which is the organizing protocol used for forest fires or oil spills (Bigley and Roberts 2001, Burkle and Hayden 2001). A full fledged incident command structure is very bureaucratic and formal; we recommend only as much formality as is necessary for the size and complexity of the response. Figure 5 shows one possible structure for organizing a response, with representative lines of responsibility and communication. The responsible host agency must decide the extent to which this model is to be followed and the degree to which it

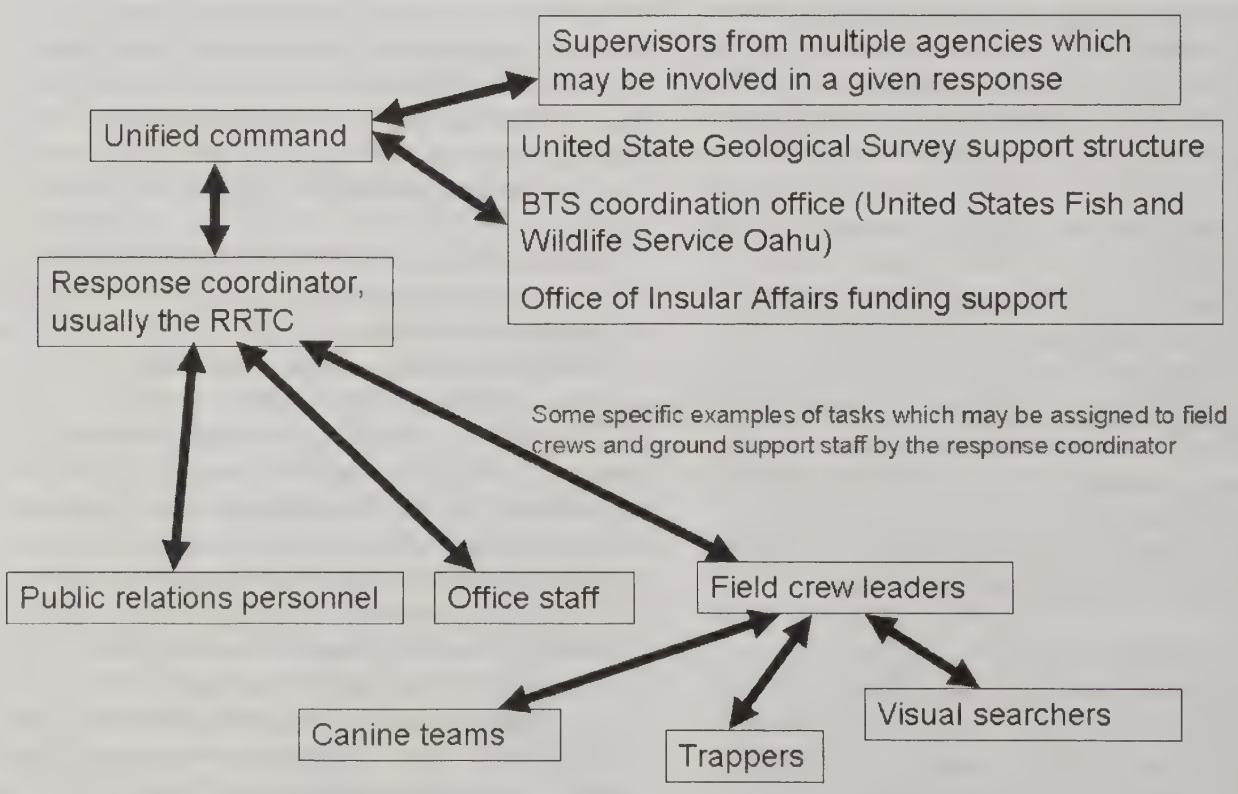



**Figure 5.** A basic brown treesnake Rapid Response Team command structure.

response, but procedures should be in place to notify everyone when roles are changed (distinctive hats are one literal way to do that). The RRTC is versed in the complexities of incident command needs to be simplified or elaborated. It may be efficient to have people change roles during the procedures and computer-aided mapping (GIS), and can assist local officials with developing a plan appropriate to the size and intricacies of likely responses on their island. Appendix B provides an outline of what such a plan might entail

One plan element that should be addressed is the procedure for transferring personnel from their day-to-day supervision to supervision during the snake response. Who will they answer to during the response? Under what conditions/times of day/dates do they get transferred? What overtime/hazardous duty rules apply to their use in snake searching? Does anyone have a medical condition or other restriction on their duties that should be conveyed to their temporary snake search supervisor? If they have concerns, to whom do they complain? Are there issues of vehicle use by temporary detailees (who can drive/ride in which vehicles)? Do volunteer forms need to be completed? Who will keep track of their restrictions/duties/hours during a snake search? To whom is information owed? For example, if a crew leader has completed a search of a specified sector, does he/she notify his/her regular supervisor, temporary search supervisor, the RRTC, or all of the above? Parallel or optional lines of responsibility are best avoided, though communication redundancy can be useful for assuring that essential knowledge is successfully transferred. For responses involving the RRT, we recommend the establishment of a unified command between the host agency and the response team as an available avenue for ensuring appropriate supervision (Figure 5).

Even with a unified command, we believe it is best to have a single overall coordinator at any one time at the field site. This eliminates the potential for individuals to receive conflicting orders or information from multiple sources. Given that response personnel are often from multiple agencies and offices within agencies, the likelihood of conflicting direction is high if the established structure is not adhered to. During a response, high motivation, a sense of urgency and a genuine desire to get the job done can cause both inexperienced and experienced personnel and managers to take unnecessary risks or to step outside of established protocols. But in the end, following established command lines will increase accountability, improve information flow, help coordination, increase operational safety, and optimize use of resources.

For small searches, it may be practical for the response coordinator to know each searcher individually and keep each person's work rules in mind. Large searches may generate a need for written records and easily-viewed credentials such as distinctive clothing or name badges. While it is simplest if each searcher is responsible for their own equipment and vehicles, a need for expensive equipment such as GPS and high-output headlamps (Lardner et al. 2007) may justify a tracking system for high-value equipment as well as all personnel.

Another central element of a snake search incident plan is geographic tracking of areas searched, areas for which entry permission has been granted, time/date searched, personnel arrivals and departures, and so forth. The incident command literature is well developed with regard to forms for tracking activities; examples are given in Appendix C.

Media coverage is valuable during a response. The media thrive on immediacy, and a snake on the loose is a hook that opens readers' minds to new information. If off-island personnel are part of the response, that fact alone will communicate to local residents the importance of reporting snake sightings. Rapid response to a snake sighting report increases the potential for media coverage through the demonstration of the urgency with which this threat is addressed. It is also wise during a response action to ask local residents to provide assistance by reporting any additional snake sightings quickly and allowing searchers to search on their properties.

A response plan should also outline the range of activities that are anticipated, the triggers for invoking that activity, and the mechanisms for ensuring their completion. Examples include night visual searches of the response area, canine team searches (night or day), barrier erection, use of ejectants and rodenticides, trapping, spotlighting, mapping, etc. Naturally, the priority assigned to any given task will be reevaluated daily during a response, but it is much easier to set in motion planned activities than to generate an organized execution spontaneously. Daily briefing can be used to verify that assignments are being completed, to assign new tasks or re-assign tasks as needed, and to provide a general update to the response personnel.

## CONCLUSION

We see four lines of defense protecting a recipient island from BTS: (1) reduction of populations on Guam, the source island, (2) interdiction efforts at Guam ports of exit, (3) interdiction efforts at recipient island ports of entry, and (4) off-port eradication of incipient populations. The latter two activities are the responsibility of the recipient island government, and for those jurisdictions that have the wherewithal, local efforts may suffice. For example, Hawaii agencies conduct virtually all of their own off-port eradication efforts, relying on the federal government primarily for assistance in training their staff on Guam. The Northern Mariana Islands have made great strides in that direction, though as a financially-strained territory they are eligible for considerable financial aid as well as support from the RRT for training and periodic direct assistance. Other island governments in the Pacific are unlikely to be able to maintain large staff of trained personnel for off-port eradication (Figure 4). Thus, we anticipate that as WS's already tight interdiction net is improved and Guam's snake population is suppressed or eradicated, the RRT will have a progressively reduced need for deploying Guam-based searchers, but will have a continued role in outreach, technology transfer, and training recipient island personnel on Guam.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

BIGLEY, G. A. AND K. H. ROBERTS. 2001. The incident command system: high reliability organizing for complex and volatile task environments. Academy of Management Journal 44:1281-1300.

BURKLE, F. M. AND R. HAYDEN. 2001. The concept of assisted management of large-scale disasters by horizontal organizations. Prehospital and Disaster Medicine 16:87-96.

BURNETT, K. M. 2007. Optimal prevention and control of invasive species: the case of the brown treesnake. Ph.D. dissertation, University of Hawaii, Manoa Hawaii, USA.

BURNETT, K. M., B. A. KAISER, B. A. PITAFI, AND J. A. ROUMASSET. 2006. Prevention, eradication, and containment of invasive species: illustrations from Hawaii. Agricultural and Resources Economics Review 35:63-77.

CAMPBELL, E. W., III. 1991. The effect of introduced roof rats on bird diversity of antillean cays. Journal of Field Ornithology 62:343-348.

CHRISTY, M. T., J. A. SAVIDGE, AND G. H. RODDA. 2007. Multiple pathways for invasion of anurans on a Pacific island. Diversity and Distributions 13:598-607.

COLVIN, B. A., M. W. FALL, L. A. FITZGERALD, AND L. L. LOOPE. 2005. Review of brown treesnake problems and control programs: report of observations and recommendations. Report to Office of Insular Affairs, Honolulu, Hawaii, USA.

ENGEMAN, R. M. AND D. S. VICE. 2002. Objectives and integrated approaches for the control of brown treesnakes. Integrated Pest Management Reviews 6:59-76.

FRITTS, T. H. 1987. Movements of snakes via cargo in the Pacific region. 'Elepaio 47:17-18.

FRITTS, T. H. 1988. The brown treesnake, *Boiga irregularis*, a threat to Pacific islands. U.S. Fish and Wildlife Service, Biological Report 88(31), Washington, D.C., USA.

FRITTS, T. H. 2002. Economic costs of electrical system instability and power outages caused by snakes on the island of Guam. International Biodeterioration and Biodegradation 49:93-100.

FRITTS, T. H. AND M. J. MCCOID. 1991. Predation by the brown treesnake on poultry and other domesticated animals in Guam. The Snake 23:75-80.

FRITTS, T. H., M. J. MCCOID, AND D. M. GOMEZ. 1999. Dispersal of snakes to extralimital islands: incidents of the brown treesnake, *Boiga irregularis*, dispersing to islands in ships and aircraft. Pages 209-223 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

FRITTS, T. H., M. J. MCCOID, AND R. L. HADDOCK. 1990. Risks to infants on Guam from bites of the brown treesnake (*Boiga irregularis*). American Journal of Tropical Medicine and Hygiene 42:607-611.

FRITTS, T. H. AND G. H. RODDA. 1998. The role of introduced species in the degradation of island ecosystems: a case history of Guam. Annual Review of Ecology and Systematics 29:113-140.

GRAGG, J. E., G. H. RODDA, J. A. SAVIDGE, G. C. WHITE, K. DEAN-BRADLEY, AND A. R. ELLINGSON. In press. Response of brown treesnakes to reduction of their rodent prey. Journal of Wildlife Management.

HAWLEY, N. B. 2007. Custom trucks, radio snake jingles, and temporary tattoos: an overview of a successful public awareness campaign related to brown treesnakes in the Commonwealth of the Northern Mariana Islands. Pages 53-56 *in* G. W. Witmer, W. C. Pitt, and K. A. Fagerstone, editors.

Proceedings of the managing vertebrate invasive species. August 2007. Fort Collins, Colorado, USA.

LARDNER, B., J. A. SAVIDGE, AND G. H. RODDA. 2007. Spotting cryptic animals in the dark: what light properties should a good headlamp have? Pages 234-245 *in* G. W. Witmer, W. C. Pitt, and K. A. Fagerstone, editors. Proceedings of the managing vertebrate invasive species. August 2007. Fort Collins, Colorado, USA.

JENKINS, P. T. 1996. Free trade and exotic species introductions. Conservation Biology 10:300-302.

MARTIN, C. 2007. Promoting awareness, knowledge and good intentions. Pages 57-62 *in* G. W. Witmer, W. C. Pitt, and K. A. Fagerstone, editors. Proceedings of the managing vertebrate invasive species. August 2007. Fort Collins, Colorado, USA.

MCCOID, M. J., T. H. FRITTS, AND E. W. CAMPBELL, III. 1994. A brown treesnake (Colubridae: *Boiga irregularis*) sighting in Texas. Texas Journal of Science 46:365-368.

MCCOID, M. J. AND D. W. STINSON. 1991. Recent snake sightings in the Mariana Islands. 'Elepaio 51:36-37.

PERRY, G. AND J. M. MORTON. 1999. Regeneration rates of the woody vegetation of Guam's northwest field following major disturbance: land use patterns, feral ungulates, and cascading effects of the brown treesnake. Micronesica 32:125-142.

PIMENTEL, D., R. ZUNIGA, AND D. MORRISON. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52:273-288.

RODDA, G. H., K. DEAN-BRADLEY AND C. P. QUALLS. 2001. The effect of prey density on brown treesnake capture probability. Micronesica 37:181-182.

RODDA, G. H. AND T. H. FRITTS. 1992. The impact of the introduction of the brown treesnake, *Boiga irregularis*, on Guam's lizards. Journal of Herpetology 26:166-174.

RODDA, G. H., T. H. FRITTS, AND E. W. CAMPBELL, III. 1999a. The feasibility of controlling the Brown Treesnake in small plots. Pages 468-477 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the Habu and the Brown Treesnake. Cornell Univ. Press, Ithaca, New York.

RODDA, G. H., T. H. FRITTS, M. J. MCCOID, AND E. W. CAMPBELL, III. 1999b. An overview of the biology of the brown treesnake, *Boiga irregularis*, a costly introduced pest on Pacific Islands. Pages 44-80 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

RODDA, G. H., T. H. FRITTS, G. PERRY, AND E. W. CAMPBELL, III. 1998. Managing island biotas: can indigenous species be protected from introduced predators such as the brown treesnake? Pages 95-108 *in* K. G. Wadsworth, editor. American Wildlife Natural Resources Conference Wildlife Management Institute, Washington, D.C., USA.

RODDA, G. H., M. J. MCCOID, T. H. FRITTS, AND E. W. CAMPBELL, III. 1999c. Population trends and limiting factors in *Boiga irregularis*. Pages 236-253 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

RODDA, G. H., J. A. SAVIDGE, C. L. TYRRELL, M. T. CHRISTY, AND A. R. ELLINGSON. 2007. Size bias in visual searching and trapping of brown treesnakes on Guam. Journal of Wildlife Management 71:656-661.

SAVIDGE, J. A. 1987. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660-668.

SIMBERLOFF, D., I. M. PARKER, AND P. N. WINDLE. 2005. Introduced species policy, management, and future research needs. Frontiers in Ecology and the Environment 3:12-20.

VICE, D. S., LINNELL, M. A., AND PITZLER, M. E. 2005. Draft summary of Guam's outbound cargo handling process: preventing the spread of the brown treesnake. U.S. Department of Agriculture, Animal Plant Health Inspect. Service, Wildlife Service, Barrigada, Guam.

VICE, D. S. AND D. L. VICE. 2004. Characteristics of brown treesnakes removed from Guam's transportation network. Pacific Conservation Biology 10:216-221.

WIEWEL, A. S., G. H. RODDA AND A. A. YACKEL ADAMS. 2006. Introduced rodent and shrew densities in the Mariana Islands: implications for brown treesnake control and management. Page 213 *in* The Wildlife Society annual meeting Anchorage, Alaska, USA.

WILCOVE, D. S., D. ROTHSTEIN, J. DUBOW, A. PHILLIPS, AND E. LOSOS. 1998. Quantifying threats to imperiled species in the United States: assessing the relative importance of habitat destruction, alien species, pollution, overexploitation, and disease. BioScience 48:607-615.

Appendix A

# USGS Snake Sighting Interview Form

2004 Edition

Haldre S. Rogers

# Interview instructions

## How to perform the interview

**Location**

- The interview should be conducted at the sighting location shortly after the sighting. If it is not possible to conduct the interview at the site, the interview should be conducted elsewhere and the location noted on Form B. In any case where the interview does not occur at the site, the interviewer should schedule a visit to the site with the observer as soon as possible.

**Observer**

- The interviewer should interview in person with the observer. However, if the interviewer cannot speak directly with the observer for language or availability reasons, the interview should occur using a third party or over the phone.

- If more than one person saw the snake, each person must be interviewed separately.

**Form**

- This packet consists of 2 forms: Form A, which is to be used during the interview with the observer and Form B, which is to be filled out by the interviewer after the interview is complete.
- Begin by entering the date of the interview on the top of each page of Form A and Form B, because this date will be the identifying number linking the pages. If more than interview is performed in one day, please enter a letter of the alphabet beginning with "a" after the date to distinguish the two sightings.
- During the interview with the observer (Form A), read the instructions for each question written in italics, then ask each question as written in bold.

- Faithfully record the observer's answers. If the observer volunteers any extra information for a question, make a note of that in the margin or on the back of the interview form

- Enter N/A for questions that are not applicable or not available.
- Some questions refer to props such as rope or cards. These are part of a prop packet. If you obtained a hard copy of this form, you should have received the prop packet with the form. If you downloaded the form from the internet, please read the additional instructions on creating a prop packet on the website.

## Interview tactics

The interview of the observer is one of the most important steps in the sighting response process. Through the interview, we are able to judge the validity of a sighting and respond accordingly. Here are some guidelines for a successful interview:

Make your interview…

- Timely: It is important to interview as soon after the sighting as possible, because memories of an event change over time and retelling of the event. We must get the information from the observer when it is still fresh in his/her head.
- Relaxed: People tend to share information more freely when they are comfortable, so anything you can do to make the interview a relaxed and welcoming occasion will improve the quality of the interview.
- Smooth: Before you perform an interview, familiarize yourself with the questions on the forms.

## Killing and Preserving Live Specimens

If the snake is alive, please kill it by freezing (preferable method) or severing the head. The specimen should be preserved as a reference or for future research. If it is impractical to store the specimen in a freezer, try to preserve the remains in a preserving fluid such as 10% formalin, 70% grain alcohol, denatured alcohol, rubbing alcohol, or rum or other high proof (> 80 proof) spirits (in order of preference). If none of these is available, the head may be dried in a slightly warmed oven, or packed in desiccating powder (such as "dri-rite"). Arrange for identification by proper authorities through the rapid response team coordinator.

| A |
|---|

## Snake Sighting Interview Form

Interviewer name: _____

Interview location: _____

**Introduction**

| | Question | Answer |
|---|---|---|
| 1 | **As you probably know, I'd like to talk to you about the snake you saw recently. First of all, thank you for reporting it. Information about this sighting may be very important to the welfare of our island. Do you have half an hour so that I could ask you some questions about this sighting?** | ☐ yes ☐ no |

*If the answer is "yes", continue on to question 2; if it is "no", reschedule the interview with the observer.*

**Name**

| | | |
|---|---|---|
| 2 | **Please spell your first name, your middle name and then your last name out loud.** | |

**Observer Contact information:**

| | | |
|---|---|---|
| 3 | **I would like to get your contact information so that we can get in touch with you if any questions come up in the future regarding this sighting. I will not use your contact information for other purposes, without your permission. May I continue?** | ☐ yes ☐ no |
| *If the answer to question 3 is "yes", continue on to question 4; if it is "no", skip to question 7.* | | |
| 4 | **What is your address?** | Address: |
| 5 | **What are your daytime and evening phone numbers?** | Daytime:<br>Evening: |
| 6 | **What is your email address, if you have one?** | ☐ My email address is: _____<br>☐ I do not have an email address |

**Observer's story**

**Please describe to me how you saw the snake. Tell me as many details as you can remember.** (write details on back on sheet)

## Present location of snake:

| | | |
|---|---|---|
| **Now, I will ask you a series of questions, some of which may be repetitive. Please bear with me if I ask you a question for which you have already given an answer because we must ask the same set of questions to each person who reports a snake. If at any point, you do not know an answer to a question, please say "I don't know".** | | |
| 7 | **Was the snake killed, captured or did it get away?** | ☐ Killed<br>☐ Captured<br>☐ It got away<br>☐ Other_____ |
| *If the answer to question 7 is "a-killed", "b-captured" or "d-other", continue on to question 8; if it is "c-it got away", skip to question 10.* | | |
| 8 | **Where is the snake now?** | |
| *If the snake is in the observer's custody, collect the snake and continue to question 9. If it is not in the observer's custody, skip to question 10, complete the interview, then make every effort to obtain the specimen.* | | |
| 9 | *Interviewer: What species is the snake?* | |
| *If the snake is a blind snake, discontinue the interview. If it is another species or unknown, continue the interview, disregarding the 'sighting information', 'snake description' and 'snake lineup' sections. Read the instructions on how to kill and/or preserve the snake on page 1.* | | |

## Time and Location of Sighting

| | | |
|---|---|---|
| 10 | **What day did you see the snake?** | _____<br>_____/_____/____<br>Day of Week<br>Day Month Year |
| 11 | **What time did you see the snake?** | Use a 24-hour clock. __ __:__ __ |
| 12 | *If necessary, ask the observer:* **In which state or on which island did you see the snake?** | |
| 13 | **What is the name of the town or region where you saw the snake?** | |
| 14 | *Detach the last page of this form (blank) and provide it to the observer with a pen.* **Please draw a map showing where you saw the snake on this piece of paper. Include all major roads and obvious landmarks in the area, so that someone who is not familiar with this island could find the place where you saw the snake. Put a star on your map at the sighting location.** | |

## Sighting Information

| | | |
|---|---|---|
| 15 | **How far away was the snake when you initially spotted it? Was it within arm's reach, closer than the length of a standard sedan, such as a Nissan Sentra, further than a sedan's length, but closer than a bus's length or further than a bus's length?** | ☐ Within arm's reach (<1m)<br>☐ Closer than the length of a standard sedan, e.g. Nissan Sentra (1m - 4m)<br>☐ Further than a sedan's length, but closer than a bus's length (4m - 10m)<br>☐ Further than a bus's length (>10 m)<br>How far? _____<br>☐ I don't know |
| 16 | **Did you get any closer to the snake?** | ☐ yes ☐ no ☐ I don't know |
| *If the answer to question 16 is "yes", continue on to question 17; if it is "no" or "I don't know", skip to question 18.* | | |
| 17 | **How close did you get to the snake? Did you get within arm's reach, closer than the length of a standard sedan, such as a Nissan Sentra, further than a sedan's length, but closer than a bus's length or further than a bus's length?** | ☐ Within arm's reach (<1m)<br>☐ Closer than the length of a standard sedan, e.g. Nissan Sentra (1m - 4m)<br>☐ Further than a sedan's length, but closer than a bus's length (4m - 10m)<br>☐ Further than a bus's length (>10 m)<br>How far? _____<br>☐ I don't know |
| 18 | **How long did you have the snake in view?** | ☐ A few seconds<br>☐ More than a few seconds but less a minute<br>☐ More than one minute but less than five minutes<br>☐ More than 5 min (how long? _____)<br>☐ I don't know |
| 19 | **On a scale of 0 to 10, with 0 being "it definitely was not a snake" and 10 being "it definitely was a snake", how confident are you that what you saw was a snake?** | |

## Snake Description

**In this section, I am going to ask you a number of questions about the specific snake that you saw. Please try to remember as many details as you can. This is probably the most crucial section of this interview, because your answers will help us to identify the animal that you saw.**

| | | |
|---|---|---|
| 20 | **Before I start on the specific questions about the description I'd like to ask you about your overall impression of the snake. Was there anything about this snake that stood out in your mind as being remarkable?** | |
| 21 | **If a snake is divided into three parts, the head, the body and the tail, what segment or segments did you see?** | ☐ Head<br>☐ Body<br>☐ Tail<br>☐ I don't know |
| 22 | *Hand the selection of ropes and PVC pipes to the observer.* **Which of these is the same thickness as the snake you saw?** | ☐ <1/4-inch<br>☐ 1/4-inch (small rope)<br>☐ 1/2-inch (medium rope)<br>☐ 3/4-inch (large rope) |
| 22 | **Can I measure the circumference of your** _____ *(fill in body part chosen in question 21)?* *If the observer answers "a"to this question, measure the body part using calipers (if available) or a tape measure (in metric units, if possible). If he/she answers "b" or "c", continue on to question 23.* | ☐ Yes: _____mm<br>☐ No<br>☐ I don't know |
| 23 | *Use the rope selected in Question 22. If the observer chose '<1/4-inch' or anything larger than 3/4-inch, use the rope with the closest diameter.* **Now we will use this rope to determine the length of the snake you saw. The black tape on the end represents the head of the snake. Feel free to manipulate the rope into the position of the snake if that will help you to determine the length.** *If the observer chooses a length that is between two colors of tape, mark the longer value in the answer column.* | ☐ Red (<0.5 m)<br>☐ Orange (0.8m)<br>☐ Black (1.0m)<br>☐ Green (1.5m)<br>☐ Blue (2.0m)<br>☐ White (2.5m or above) |

| | | |
|---|---|---|
| 24 | *Spread the markings cards in front of the observer.* **Choose the card with a pattern that most closely matches the markings on the snake you saw.** | ☐ Solid pattern<br>☐ Colored head<br>☐ Linear multicolor<br>☐ Spotted<br>☐ Banded<br>☐ Striped<br>☐ Blotched<br>☐ Speckled<br>☐ I don't know |
| 25 | *Hand the observer a color wheel. Record the numbers associated with the color chosen by the observer.* **Please choose the color that most closely resembles the main color of the snake. You may use either side of the color wheel.** | |
| 26 | **What part of the snake was that color?** | |
| 27 | **Did you see any other colors on the snake?** | ☐ yes ☐ no ☐ I don't know |
| *If the answer to question 27 is "yes", continue on to question 28; if it is "no" or "I don't know", skip to question 29.* | | |
| 28 | **What other colors did you see on the snake? Please tell me where on the snake these colors were.** | Color _____<br>Location _____<br>Color _____<br>Location _____ |
| 29 | **What was the sheen of the snake? Was it glossy like patent-leather shoes, semi-glossy like ordinary cowhide leather or flat like suede leather?** | ☐ Glossy (like patent-leather shoes)<br>☐ Semi-glossy (like ordinary cowhide leather)<br>☐ Flat (like suede leather)<br>☐ I don't know |
| 30 | *Spread all head shape cards on the table in front of observer.* **Please select the head shape that most closely resembles the head of the snake you saw.** | ☐ Broad head<br>☐ Medium head<br>☐ Narrow head<br>☐ I don't know |
| 31 | *Use pupil cards. Omit this question if observer did not see the head of the snake (refer to question 21).* **Were the snake's pupils elliptical or round**? | ☐ Elliptical<br>☐ Round<br>☐ I don't know |

## Snake Behavior

| | | |
|---|---|---|
| 32 | **Was the snake moving or still when you first saw it?** | ☐ Still<br>☐ Moving (where? _____)<br>☐ Other _____ |
| *If the answer to question 32 is "a- still", skip to question 34. If it is "b- moving" or "c- other', continue on to question 33.* | | |
| 33 | **At what speed was the snake traveling when you first saw it? Was it the speed of a slow walk, a normal walk, a fast walk or a run?** | ☐ Slow walk<br>☐ Normal walk<br>☐ Fast walk<br>☐ Run |
| 34 | *Show the posture cards to the observer one by one.* **Which picture best illustrates the snake's posture when you first saw it?** | ☐ Coiled<br>☐ Looped<br>☐ Sinuous<br>☐ Defensive "S"<br>☐ Cobra<br>☐ I don't know |
| 35 | **Did you see the snake exhibit any of these other postures? Choose all that apply.** | ☐ Coiled<br>☐ Looped<br>☐ Sinuous<br>☐ Defensive "S"<br>☐ Cobra<br>☐ No<br>☐ I don't know |

**Snake Line-up**

| | | |
|---|---|---|
| 36 | *Ask the question written below, then, one by one, in numerical order, show the pictures of snakes. If the observer chooses a picture, record the number on the back of the card along with any remarks about the picture (i.e. "the head looked like this one"). Go through pictures a second time if requested by the observer.* **Please select any picture that looks like the snake you saw. You may choose more than one picture. The snake you saw may not be in this assortment of pictures, so I will go through these once, during which time you may choose pictures of snakes that look similar to the one you saw. If you have not chosen any pictures once I have finished showing all of the options, I will go through the options a second time. If the snake does not resemble any of these options, do not choose any** *(interviewer: mark "none" in that case).* | Card Number<br>Comments<br><br><br>

## Sighting Location Details

| | | |
|---|---|---|
| 37 | **Was the snake in a tree, on the ground or in water?** | ☐ Tree<br>☐ Ground<br>☐ Water<br>☐ Other _____ |
| 38 | **Was the snake on the road, on the roadside, in the jungle, in a grassy area, amongst buildings or somewhere else?** | ☐ Road<br>☐ Roadside<br>☐ Jungle<br>☐ Grassy area<br>☐ Residential area<br>☐ Other |
| 39 | **How high was the vegetation at the exact spot of the snake sighting? Was it bare ground, mowed grass, ankle high, mid-calf, knee-high, waist-high, head-high, as high as you can reach or tall jungle?** | ☐ Bare ground (0 mm)<br>☐ Mowed grass (1-80 mm)<br>☐ Ankle high (81-190 mm)<br>☐ Mid-calf (191-400 mm)<br>☐ Knee-high (401-600 mm)<br>☐ Waist-high (601mm-1.3m)<br>☐ Head-high (1.4m-2.0m)<br>☐ As high as one can reach (2.1m-2.6m)<br>☐ Tall jungle (>2.6m)<br>☐ I don't know |
| 40 | **What was in the 3 meters around where you saw the snake? I am going to read out some options. Say yes or no to each option. Were there buildings, vehicles, yard vegetation, farm vegetation, jungle, grassy area or pavement?** | ☐ Buildings<br>☐ Vehicles<br>☐ Yard vegetation<br>☐ Farm vegetation<br>☐ Jungle<br>☐ Grassy area<br>☐ Pavement<br>☐ Other _____ |

## Weather

| | | |
|---|---|---|
| 41 | **To the best of your knowledge, did it rain in the 6 hours prior to the sighting?** | ☐ yes ☐ no ☐ I don't know |
| 42 | **Was it raining at the time of the sighting?** | ☐ yes ☐ no ☐ I don't know |
| *If the answer to question 42 is "yes", continue on to question 43. If it is "no" or "I don't know", skip to question 44.* | | |
| 43 | **How hard was it raining? Was it a misty fog, drizzle, light rain, heavy rain or torrential rain?** | ☐ Mist fog<br>☐ Drizzle<br>☐ Light rain<br>☐ Heavy rain<br>☐ Torrential rain |
| 44 | **What was the amount of natural light on the snake at the time of the sighting? Was it direct sunlight, during the day without direct sunlight, at dawn or dusk where natural lighting was low but sufficient to see color, under a full moon or on a dark night?** | ☐ direct sunlight<br>☐ during the day, but without direct sunlight<br>☐ at dawn or dusk where natural lighting was low but sufficient to see color<br>☐ full moon<br>☐ dark night |
| *If the answer to question 44 is "a- direct sunlight" or "b- during the day, but without direct sunlight", continue on to question 45. If it is "c- at dawn or dusk", "d- full moon" or "e- dark night", skip to question 46.* | | |
| 45 | *Ask this question only if the observer saw the snake during the day (use your local knowledge of lighting conditions for the time answered in question 6 to determine if it was during the day or night).* **Was it sunny, partly cloudy or overcast when you saw the snake?** | ☐ Sunny<br>☐ Partly cloudy<br>☐ Overcast<br>☐ I don't know |
| 46 | **Was there an artificial source of light, such as streetlights, car headlights or a flashlight, when you saw the snake?** | ☐ yes ☐ no |
| *If the answer to question 46 is "yes", continue on to question 47; if it is "no", skip to question 48.* | | |
| 47 | **What was this light source?** | |

## Local information

| | | |
|---|---|---|
| 48 | **What is the approximate distance between where you saw the snake and the nearest airport or seaport?** *Have the observer estimate the distance. Accept any answer given. Be sure to record the units given by observer.* | _____ Port Name<br>_____ Distance |
| 49 | **How far is it between where you saw the snake and the ocean?** *Have the observer estimate the distance. Accept any answer given. Be sure to record the units given by observer.* | |
| 50 | **Is there any construction or cargo that has recently arrived on island near the sighting location?** | ☐ yes ☐ no ☐ I don't know |
| *If the answer to question 50 is "yes", continue on to question 51; if it is "no" or "I don't know", skip to question 53.* | | |
| 51 | **What type of construction/cargo is near the sighting location?** | |
| 52 | **Exactly where is this construction/cargo located?** | |
| 53 | **Do you know the name and phone number of the person who owns or manages the land where you saw the snake?** | Name:<br><br>Phone Number: |

## Other observers

| | | |
|---|---|---|
| 54 | **Did anyone else see this snake?** | ☐ yes ☐ no ☐ I don't know |
| *If the answer to question 54 is "yes", continue on to question 55; if it is "no" or "I don't know", skip to question 56* | | |
| 55 | **Could you give me their name, address and phone number, if you know it?** *Please gather as much and as accurate contact information as possible because often addresses in remote locations are vague or unavailable, making future contact with the observer difficult. Each observer must be interviewed separately.* | |

## Prior experience with snakes

| | | |
|---|---|---|
| **I'm going to finish up with a few questions about your previous experience with snakes.** | | |
| 56 | **Which of the following choices best describes your experience with snakes: None; in photographs, books, TV and videos; as pets or live exhibits at zoos; seeing wild snakes caught by others; or personally capturing wild snakes?** | ☐ None<br>☐ Photographs, books, TV and videos<br>☐ As pets or live exhibits at zoos<br>☐ Seeing wild snakes caught by others<br>☐ Personally capturing wild snakes |
| 57 | **Roughly how many live snakes have you handled?** | ☐ 0<br>☐ 1-5<br>☐ 5-25<br>☐ >25 |
| 58 | **What kind of snakes have you handled?** | |

## Other comments

| | | |
|---|---|---|
| 59 | *Record every detail mentioned by the observer because each detail could be important at a later date.* **Please tell me anything I have missed that you think could be important.** | |
| 60 | **Do you have any other comments?** *Record any other comments made by observer exactly as they are stated.* | |
| 61 | **Thank you for sharing this information with us. Can we share the details of your sighting with other researchers or managers?** | ☐ yes ☐ no |
| 62 | **May we include the information from this interview in a database that will be available to the general public? Your name and contact information, however, will NOT be available to the public.** | ☐ yes ☐ no |

## On-site examination

*Ask the observer to tell his/her story of the sighting. Mark the place where the observer was when he/she first saw the snake and where the snake was when the observer first spotted it.*

Your interview with the observer is now complete. Please fill out form B now.

B

# Questions for Interviewer

This form is to be filled out by the interviewer.

## Interviewer contact information

| | | |
|---|---|---|
| 1 | Name of interviewer | |
| 2 | Interviewer Address | |
| 3 | Interviewer Phone Number | daytime:<br>evening: |
| 4 | Interviewer Email | |

## Observer information

| | | |
|---|---|---|
| 5 | Name of Observer: | |
| 6 | Estimated age of observer: | |

## Chronology of Events

| | | |
|---|---|---|
| 7 | Date and time sighting was reported to officials | _____/_____/____ __ __:__<br>Day Month Year Time |
| 8 | Date and time of initial contact with observer | _____/_____/____ __ __:__<br>Day Month Year Time |
| 9 | Date and time of interview | _____/_____/____ __ __:__<br>Day Month Year Time |

## Method of interview:

| | | |
|---|---|---|
| 10 | Was the interview conducted with the observer or through a third party? | ☐ With the observer<br>☐ Through a third party |
| 11 | Was the interview conducted person-to-person or over the phone? | ☐ Person-to-person<br>☐ Over the phone |
| 12 | Was the interview conducted at the location of the sighting, at the observer's home or elsewhere? | ☐ At the location of the sighting<br>☐ At the observer's home<br>☐ Elsewhere_____ |

## Location

| | | |
|---|---|---|
| 13 | What country was the sighting in? | |
| 14 | On which island or in which state was the sighting? | |
| 15 | Did you visit the sighting location? | ☐ yes ☐ no |
| 16 | Please describe where the sighting was located, using specific landmarks and road names. | |
| 17 | Do you foresee any difficulties getting permission from the landowner to search the area? | ☐ yes ☐ no<br>Comments: |

## Interviewer's opinion of sighting

| | | |
|---|---|---|
| 18 | On a scale of 0 to 10, with 0 being "the observer definitely did not see a snake" and 10 being "the observer definitely saw a snake", how confident are you that what the observer saw was a snake? | 0 1 2 3 4 5 6 7 8 9 10<br>Not a snake<br>Definitely a snake |

## Interviewer background

| | | |
|---|---|---|
| 19 | Prior to this sighting, what was your primary experience with snakes? | ☐ none<br>☐ photographs, books, TV and videos<br>☐ as pets or live exhibits at zoos<br>☐ seeing wild snakes caught by others<br>☐ personally capturing wild snakes<br>☐ Other: _____ |
| 20 | Roughly how many live snakes have you handled? | ☐ 0<br>☐ 1-5<br>☐ 5-25<br>☐ >25 |
| 21 | What kind of snakes have you handled? | |

## On-site examination- to be completed at the location of the sighting.

| | | |
|---|---|---|
| 22 | Measure the distance between observer and snake at time of initial sighting using a metric tape measure. | |
| 23 | If snake was off the ground, measure vertical distance from ground using metric tape measure. | |
| 24 | Record the UTM coordinates for the sighting location using a GPS and the WGS-84 datum. | |
| 25 | What are the dominant vegetation types in the 1 ha area around the sighting? Please be as specific as possible, using Latin species names if known. | |
| 26 | What potential non-snake candidates for the sighting are present in the area (i.e., rats, water hoses, monitor lizards etc.)? | |
| 27 | Is there any recent construction or cargo in the area? If yes, please describe it. | Yes<br>_____<br>No |

# USGS Snake Sighting Overall Search Plan

2007 edition
Haldre S. Rogers and James W. Stanford

## Overall search plan

Response code: _____ Response Command: _____
Sighting date: _____ Sighting location: _____
Search plan created by: _____

Circle all tools that will be used in this response:

Trapping Rodenticide Night searches Daytime searches Ejectants

Canine Team Night Searches Canine Team Day Searches

Acetaminophen Temporary Barrier

Agencies and their staff involved during the response action:

## Trapping Yes No

Person in charge: _____

Total number of traps available: _____

Total number of bait available: Live mice: _____ Dead mice: _____

Other (please describe): _____

Potatoes? Yes No Feed blocks? Yes No

Date traps will be activated: _____

Date traps will be deactivated: _____

Traps will be checked @ _____ by: _____

| Segment | Number of Traps | Location of traps |
|---|---|---|
| | | |
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| | | |

**Rodenticide** **Yes** **No**

Person in charge: _____

Total number of bait stations available: _____

Total amount of rodenticide available: _____

Date bait stations will be activated: _____

Date stations will be deactivated: _____

Segment ___ Number of bait stations: _____ Location of bait stations: _____

Segment ___ Number of bait stations: _____ Location of bait stations: _____

Segment ___ Number of bait stations: _____ Location of bait stations: _____

Segment ___ Number of bait stations: _____ Location of bait stations: _____

Segment ___ Number of bait stations: _____ Location of bait stations: _____

Segment ___ Number of bait stations: _____ Location of bait stations: _____

Segment ___ Number of bait stations: _____ Location of bait stations: _____

Segment ___ Number of bait stations: _____ Location of bait stations: _____

Bait stations will be checked by: _____

Bait stations will be checked: daily every 2 days

## Night Searching Yes No

Person in charge: _____

Total number of searchers available and initials of each:

RRT _____

Local _____

Segments to cover: _____

Date night searching will begin: RRT _____ Local _____

Date night searching will end: RRT _____ Local _____

Hours spent conducting searches each night: RRT _____

Local _____

Maximum search area: _____

Day 1 (date _____): Inner 50 meters = Segments _____

Day 2 (date _____): To 100 meters = Segments _____

Day 3 (date _____): To 150 meters = Segments _____

Day 4 (date _____): To 200 meters = Segments _____

Day 5 (date _____): To 250 meters = Segments _____

Day 6 (date _____): To 300 meters = Segments _____

Day 7 (date _____): To 300 meters = Segments _____

Day 8 (date _____): To 300 meters = Segments _____

Day 9 (date _____): To 300 meters = Segments _____

Day 10 (date _____): To 300 meters = Segments _____

Day 11 (date _____): To 300 meters = Segments _____

Day 12 (date _____): To 300 meters = Segments _____

Day 13 (date _____): To 300 meters = Segments _____

Day 14 (date _____): To 300 meters = Segments _____

Day 15 (date _____): To 300 meters = Segments _____

Day 16 (date _____): To 300 meters = Segments _____

Day 17 (date _____): To 300 meters = Segments _____

Day 18 (date _____): To 300 meters = Segments _____

**Daytime searches** **Yes** **No**

Person in charge: _____

Total number of searchers available and initials:

RRT _____

Local _____

Segments to cover: _____

Date daytime searching will begin: RRT _____ Local _____

Date daytime searching will end: RRT _____ Local _____

Hours spent conducting search each day: _____

Targets: Pandanus ground burrows other (list): _____

**Ejectants** **Yes** **No**

Person in charge: _____

Chemical available: CloveCinnamon Hair Spray Other: _____

Total number of people available and initials:

RRT _____

Local _____

Segments to cover: _____

Date repellent use will begin: _____

Date repellent use will end: _____

Hours spent using repellents each day: _____

Targets: Cliff holes Rodent Burrows Pipes Other: _____

## Canine teams Yes No

Person in charge: _____

Canine teams used (list dog, handler, affiliations, and whether forest or cargo trained):

_____

_____

_____

Segments to cover: _____

Date dog searches will begin: _____

Date dog searches will end: _____

Hours spent performing dog searches each day: _____

Targets: _____

Day 1 (date _____): Segments (by team) _____

Day 2 (date _____): Segments _____

Day 3 (date _____): Segments _____

Day 4 (date _____): Segments _____

Day 5 (date _____): Segments _____

Day 6 (date _____): Segments _____

Day 7 (date _____): Segments _____

Day 8 (date _____): Segments _____

Day 9 (date _____): Segments _____

Day 10 (date _____): Segments _____

Day 11 (date _____): Segments _____

Day 12 (date _____): Segments _____

Day 13 (date _____): Segments _____

Day 14 (date _____): Segments _____

## Acetaminophen Yes No

Person in charge: _____

Total number of bait tubes available: _____

Total number of dead mice available: _____

Total amount of acetaminophen available: _____

Segments to cover: _____

Date bait tubes will be baited: _____

Date tubes will be deactivated: _____

Segment ___ Number of bait tubes: _____ Location of bait tubes: _____

Segment ___ Number of bait tubes: _____ Location of bait tubes: _____

Segment ___ Number of bait tubes: _____ Location of bait tubes: _____

Segment ___ Number of bait tubes: _____ Location of bait tubes: _____

Segment ___ Number of bait tubes: _____ Location of bait tubes: _____

Segment ___ Number of bait tubes: _____ Location of bait tubes: _____

Segment ___ Number of bait tubes: _____ Location of bait tubes: _____

Segment ___ Number of bait tubes: _____ Location of bait tubes: _____

Bait tubes will be checked by: _____

Bait tubes will be checked: daily every 2 days weekly

## Temporary Barrier Yes No

Person in charge: _____

Available on island? Yes No

Circumference of area to be enclosed by barrier: _____ meters

Number of people available to set up: RRT _____

Local _____

Length of shade cloth available: _____

Sand to be collected from: _____

Date to be erected: _____

Date to be removed: _____

Landowner permission to clear branches/trees? Yes N

## Other Tools Yes No

Person in charge: _____

Describe Tool and Use:

Appendix C

# USGS Snake Sighting Response Data Sheets

2007 edition

## Saipan Search Data

| | | | |
|---|---|---|---|
| Primary Searcher | | | |
| Other Searchers | | | |
| Date | | | |
| Page | | of | |
| Location | | | |
| Arrival time | | | |
| Clouds (%) | | | |
| Rain Past (y/n) | | | |
| Rain Now (y/n) | | | |
| Wind (Beaufort #) | | | |
| Gust (Beaufort #) | | | |
| Wind Direction | | | |
| Moon | | | |
| Light Source | | | |

| Search Type | Dist (M) | Segment | Start | Stop |
|---|---|---|---|---|
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## Non-Snake Perch Data

| Time | Species | Vert (M) | Perch Dia (mm) | Perchtax |
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Trapper Date

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# FLOTATION MATERIALS FOR AERIAL DELIVERY OF ACETAMINOPHEN TOXIC BAITS TO BROWN TREESNAKES

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***Abstract:*** Polyvinyl chloride (PVC) tubes are effective bait stations for delivering dead neonatal mice (DNM) treated with the oral toxicant, 80 mg acetaminophen, to brown treesnakes *(Boiga irregularis)* in accessible jungle forest on Guam. However, PVC tubes are not practical for delivery of baits to remote areas of jungle or the forest canopy. Further, it is important that baits entangle in the canopy and not fall to the ground where they can be scavenged by non-target animals such as crabs. Data from helicopter aerial deployment of untreated DNM with radio transmitters that landed on the ground in areas of high coconut crab *(Birgus latro)* and hermit crab *(Coenobita* spp.*)* abundance showed that 67% of DNM were taken by crabs and 11% by monitor lizards *(Varanus indicus)*. In contrast, in low crab abundance areas crabs took 24% of the DNM that landed on the ground. It is evident from these data that a flotation system that delivers DNM to the canopy is needed; otherwise non-target animals will remove DNM, making them unavailable for snakes. Seven aerial flotation devices were evaluated. Promising aerial devices are two types of commercial cardboard paper streamers that resulted in 75% - 85% of the DNM becoming entangled in the canopy.

***Key Words:*** aerial delivery, bait, *Boiga irregularis,* brown treesnake, crabs, Guam, invasive species, parachute, radiotelemetry.

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## INTRODUCTION

The brown treesnake *(Boiga irregularis)* is an invasive exotic nocturnal predator that was probably introduced on the snake-free island of Guam as a stowaway in cargo in the late 1940s (Fritts 1988, Rodda et al. 1992). In the absence of natural predators and an abundant food supply the snake colonized the entire island with population densities in some areas as high as 50-100 snakes per hectare (Rodda et al. 1992, Rodda et al. 1999). The brown tree snake is considered to be the primary factor responsible for the decline and extirpation of many native birds and lizards on Guam (Savidge 1987, Fritts 1988, Rodda and Fritts 1992, Drahos 2002) and the decline of fruit bats *(Pteropus mariannus)* (Wiles 1987). It is mildly venomous and poses a health risk to small children (Fritts et al. 1994), causes power outages by climbing on electrical transmission wires (Fritts et al. 1987), and preys on poultry and domesticated birds (Fritts and McCoid 1991). Guam is the focal point of air and ship cargo traffic in the northern western Pacific and there is a threat that snakes could be inadvertently introduced and establish breeding populations on other snake-free islands (e.g., Hawaii) in the Pacific region (Fritts et al. 1999). In 1993 the United States (US) Department of Agriculture (USDA), Wildlife Services, initiated an operational program to deter the spread of snakes from Guam using hand capture from fences during nighttime spotlight searches, trapping, and inspection of cargo with search dogs (Hall 1996, Engeman and Vice 2001, Vice and Vice 2004). However, these methods have limited use for reducing snakes in large inaccessible landscapes required for the reintroduction of endangered species (BTSCC 1996).

Dead neonatal mice (DNM) baits treated with 80 mg acetaminophen distributed in bait stations are effective for reducing brown treesnake populations in small 5-7 ha plots of accessible jungle forest on Guam (Savarie et al. 2001), but a practical system for broadcast of baits to inaccessible areas of jungle is not available. Since 2001, studies have been ongoing to develop aerial bait delivery techniques. Small plastic parachutes (Shivik et al. 2002) and paper corn starch devices (L. Clark, personal communication) hand dropped from a helicopter have been used as flotation devices for entangling dead mice in the forest canopy. Canopy entanglement was 92% for plastic parachutes and 42% for the corn starch paper. But

both have negative aspects: plastic takes at least two years to degrade, and the corn starch paper dissolves in the rain (which can occur daily on Guam), dropping the bait to the ground. Savarie and Tope (2004) evaluated 5 biodegradable flotation devices; results were marginally successful with 62% canopy entanglement with a commercial food cup.

The primary reason for placing DNM baits in the canopy is to keep them from being taken by terrestrial scavengers such as feral pigs *(Sus scrofa)*, monitor lizards *(Varanus indicus)*, coconut crabs *(Birgus latro)*, land hermit crabs *(Coenobita* spp.*)*, and marine toads *(Bufo marinus)*. Baits taken by non-targets result in fewer baits available for snakes and, therefore, a proportionately higher number would have to be delivered to maximize snake control. Three aerial bait studies (Shivik et al. 2002, Savarie and Tope 2004, L. Clark, personal communication) conducted on Andersen Air Force Base (AAFB), Guam, in areas known to have low crab densities and high densities of feral pigs revealed no evidence that feral pigs took ground-placed baits; bait-take was negligible by monitor lizards (<1%); and 6-11% for crabs and toads. However, a study in an area of high crab abundance is needed to quantify the potential of this non-target to remove baits.

Data from 3 aerial bait delivery studies conducted on Guam using untreated thawed DNM are reported in this paper: (1) comparative bait-take by BTS and non-target animals for baits delivered in areas of low and high crab abundance, (2) efficacy of paper towel and biodegradable plastic-like parachutes for entangling in the canopy, and (3) efficacy of four commercial biodegradable paper products for entangling in the canopy. The sole purpose of studies 2 and 3 was to determine the ability of the parachutes to entangle in the forest canopy and bait-take was not recorded.

## METHODS

### Bait-take in Low and High Crab Abundance Areas

Four 50 x 200 m aerial drop sites adjacent to roads were selected, 2 on AAFB, Tarague Beach Road, and 2 on US Naval Base, Orote Peninsula. Both sites are secondary growth forests. Vegetation and trees on the Tarague site have been described (Shivik et al. 2002). Tangantangan *(Leucaena leucocephala)* is the predominant tree on Orote. To assess crab abundance on these sites, 3 lure stations were established at 72-107 m intervals along the midline of each of the 4 drop sites. Stations were baited with a fermented attractant mixture of boiled rice, shaved coconut, and water (USFWS 2001) with pieces of raw coconut in a 2 m diameter circle on the ground. Attractant mixture was replenished at least once daily for 3 days, depending on rainfall. Lure stations were checked prior to the aerial drops for 3 consecutive nights between 2000-2400 hr and numbers and species of crabs counted.

Unattached DNM (5-7 g, Essex Co., Blum, Texas) and DNM attached to biodegradable jute netting (20.3 x 20.3 cm, 5.7 cm thick mesh) by a 30.5 cm long cotton thread to a rear leg were deployed by hand from a US Navy Knighthawk MH-60S helicopter from about 30 m above ground level (AGL). Twenty (20) DNM baits (10 unattached and 10 attached to netting) were deployed per drop on each of the 4 drop sites during August-September 2004. A radio transmitter (Model F1620, 1.9 g, Advanced Telemetry Systems [ATS], Isanti, Minnesota) was attached to the abdomen of each mouse with polyurthane adhesive (Gorilla Glue Co., Cincinnati, Ohio) and the 10.2 cm external whip antenna extended along the tail of the mouse.

Immediately after each bait drop, baits were tracked by radio receiver and visually located on the ground or in the forest canopy and height (m) in the canopy estimated. Status of the baits was checked daily for 1-7 days and radios were recovered from baits not consumed. Snakes that consumed baits were captured, euthanized, and its distance (m) from the initial bait drop point was estimated. Body mass (g), snout-vent length (SVL, mm), and sex were recorded and radio transmitters recovered.

### Biodegradable Parachutes

In September-October 2005, 2 aerial bait drops were conducted on 50 x 200 m drop zones; 1 each on AAFB, Tarague Beach Road, and US Naval Computer and Telecommunications Station Guam (NCTS, Haputo Beach Road). Vegetation on NCTS is similar to Tarague. Two types of biodegradable parachutes were evaluated: paper towel, 23.8 x 27.3 cm, A-A-696 Type 1 Singlefold, Lighthouse for the Blind, New Orleans, Louisiana; and a plastic-like material, 20.3 x 20.3 cm, EcoFilm®, Cortec Corp., St. Paul, Minnesota. Four pieces of cotton thread (3-30.5 cm long and 1-35.6 cm long) were individually tied to the corners of each parachute type. The threads were knotted and the longer thread was tied to a rear leg of a DNM.

A radio transmitter (ATS Model F1620) was glued to each DNM bait. Twenty parachutes (10 of each type of parachute with DNM) were delivered by hand from a Knighthawk MH-60S helicopter from about 60 m AGL on Tarague and from a Hughes 500 helicopter from about 30 m AGL on NCTS. Immediately after each drop, bait locations were determined by radiotelemetry and height in canopy estimated.

**Biodegradable Paper Products**

Four biodegradable commercial paper products were evaluated in August 2006: (1) single white flag, 3 m long x 10.2 cm wide paper streamer attached to cardboard on one end, (2) double white flag, 4.9 m long x 10.2 cm wide paper streamer attached to cardboard on both ends (single and double white flags from R-S Sales/Automatic Flagman®, Lewiston, Idaho), (3) paper plate, 22.9 cm diameter with daisy-like cutout pattern with intact outer edge, Wal-Mart, Bentonville, Arkansas, and (4) paper cup, 237 ml, Marfred Industries, Sun Valley, California. ATS Model R1645 radio transmitters (0.9 g) were glued to the abdomen of DNM. A rear leg of a DNM was secured to the cardboard on the single and double white flags with hot glue. Four lengths of cotton thread as described above were attached to the paper plates and cups tied to a DNM. The paper products were dropped by hand from a Hughes 500 helicopter from about 30 m AGL on a 50 x 200 m drop zone on NCTS, Haputo Beach Road. Five each of the 4 types of paper products were delivered on each of 3 drops, and 5 each of single white and double ender flags were delivered on 1 drop. Locations of baits was determined by radiotelemetry.

In each of the three studies, DNM were thawed and prepared the day before aerial delivery. They were then re-frozen overnight and thawed the next day before aerial deployment. In addition to the radio that was glued to each DNM, a 13 x 30 mm strip of pink flagging was also glued to each DNM to aid in visual identification.

As appropriate, descriptive statistics and chi-square analysis were used to analyze the data. A chi-square test was used to compare observed and predicted frequencies of bait-take in areas of low and high crab abundance. Proportions were in some cases, further examined using the chi-square subdivision method (Zar 1984). Subdivision was performed only when the initial test was declared significant ($P \leq 0.05$), and to minimize Type I error when evaluating a subdivision, significance was declared at $P \leq 0.01$.

## RESULTS

**Bait-take in Low and High Crab Abundance Areas**

After observing crab lure stations for 3 nights, no crabs were recorded on the 2 sites on Tarague, whereas on the 2 Orote sites 44 coconut crabs and 9 land hermit crabs were recorded: 22 coconut crabs and 4 hermit crabs from one site, and 22 coconut crabs and 5 hermit crabs from the second site. Based on these numbers, Tarague was classified the "low" abundance crab site and Orote the "high" abundance site.

Bait-take and fate of baits on the ground at the low and high abundance crab sites are shown in Table 1. Frequencies of bait-take differed among bait-take categories ($\chi^2 = 26.0$, $df = 3$, $P < 0.0001$). Bait-take by crabs showed the largest deviation from its expected value, indicating take by crabs contributed most to the observed significance. Therefore, the chi-square was subdivided by removing bait-take by crabs from the analysis, which again, differed among categories ($\chi^2 = 14.6$, $df = 2$, $P < 0.0007$). Here, the category "other" (combined take by monitor lizards, toads, ants, and unknown) showed the largest deviation between observed and expected.

One of the 20 baits attached to jute netting dropped on Tarague, the low crab abundance site, could not be visually located. Of the remaining 19 baits, 5 (26%) landed in the canopy at a mean height of 4 m. Four of these 5 baits detached from the jute netting and fell to the ground, 3 after 1 day and 1 after 2 days. The bait remaining in the canopy was consumed by ants after 3 days. Seven of 14 baits that landed on the ground were detached from the jute netting. All of the 20 baits without jute netting landed on the ground.

On Orote, the high crab abundance site, 15% (3 of 20) of the baits attached to jute netting entangled in the canopy at a mean height of 3 m. One of the 3 baits was taken by a crab; 1 bait remained in the canopy for 3 days, and the fate of 1 bait could not be determined. The radio transmitter detached from one of the 20 baits attached to jute netting and was immediately recovered. One bait was consumed by a monitor lizard before it could be visually inspected and 11 of the 15 baits on the ground were detached from the jute netting. All of the 20 baits without jute netting landed on the ground.

Snakes consumed baits only at the low crab abundance site. Eight snakes consumed 9 baits from the ground and 7 of the 9 baits were taken on

day 2 after the drop. The 8 snakes traveled a mean overnight distance of 29 m (range 6-95 m). Three of the 8 snakes that consumed bait were captured (2 females: 68.2 g, 826 mm SVL and 118.4 g, 990 mm SVL, and 1 male: 127.7 g, 1065 mm SVL). The other 5 snakes remained elusive, but their radios were expelled and recovered 2-7 days post-consumption. Two of the 9 baits (22%) consumed by crabs on the low crab abundance site were taken on night 1 and the remaining 7 baits on nights 2 and 3. On the high crab abundance site, crabs took 22 of 24 baits (92%) on night 1 and the remaining 2 baits on night 2.

**Biodegradable Parachutes**

There was no difference in canopy entanglement of baits between the Tarague and NCTS drop zones for the Ecofilm® and paper towel parachutes ($\chi^2$ = 3.8, $df$ = 3, $P$ = 0.28). Fifteen of 20 (75%) of the Ecofilm® parachutes (7 of 10 on Tarague and 8 of 10 on NCTS) and 17 of 20 of the paper towel parachutes (7 of 10 on Tarague and 10 of 10 on NCTS) were entangled in the canopy. Mean height in the canopy ranged from 3-5 m for each type of parachute at each of the drop zones. No baits detached from these parachutes.

**Biodegradable Paper Products**

Canopy landings for the single white flags, double white flags, paper plates, and paper cups were 85%, 95%, 67%, and 80%, respectively, and there was no difference among these four paper products for entanglement in vegetation ($\chi^2$ = 5.0, $df$ = 3, $P$ = 0.17). Mean height in canopy ranged from 2-4 m for each of the four paper products. Of the baits that landed in the canopy, 1 of 17 (6%) detached from single white flags and 2 of 19 (11%) detached from double white flags. No baits detached from either the paper cups or paper plates.

## DISCUSSION

Based on results from the crab lure stations, there is a dramatic difference in numbers of crabs between Tarague and Orote. Anecdotal information indicates that high feral pig numbers limit crab populations (USFWS 2001). Signs of pigs, including visual sightings and ground rootings, were prevalent on Tarague, but no signs of pigs were observed on Orote.

Although no crabs were recorded on Tarague from the crab lure stations, a hermit crab was observed with 1 of the 9 baits designated as being taken by crabs (Table 1). A hermit crab was also observed with 1 of the 24 baits taken by crabs on Orote. Evidence that crabs took baits included damage to the radio antenna or battery. And radios with or without the dead mouse bait, or a partially shredded mouse bait, were often located in a protected crab refugium under logs, stumps, or rocks. Characteristically, crabs would consume the flesh of the mouse leaving the radio attached to the glue.

**Table 1.** Bait-take and fate of baits on ground at low (Tarague) and high (Orote) crab abundance sites.

| Bait-take/ fate of bait | Low crab abundance (Tarague) | | High crab abundance (Orote) | |
|---|---|---|---|---|
| Snake | 9*/38 | 23.7% | 0/36 | 0.0% |
| Crab | 9/38 | 23.7% | 24/36 | 66.7% |
| Monitor lizard** | 0/38 | 0.0% | 4/36 | 11.1% |
| Marine toad** | 1/38 | 2.6% | 0/36 | 0.0% |
| Ants** | 8/38 | 21.1% | 5/36 | 13.9% |
| Unknown** | 1/38 | 2.6% | 3/36 | 8.3% |
| No take | 10/38 | 26.3% | 0/36 | 0.0% |
| Totals | 38/38 | 100% | 36/36 | 100% |

* 8 snakes consumed 9 baits.

** For purposes of chi-square analysis, these 4 entities were grouped in the "other" category.

The impact of crabs as major scavengers of baits for snakes is indisputable and not surprising (Table 1). Three times more baits were taken by crabs on Orote (67%) as compared to Tarague (24%). And the rapid disappearance of baits by crabs on Orote (92% of baits taken on night 1) indicates that baits would not be available during night 2 when they are most attractive to snakes (Jojola-Elverum et al. 2001). Second to crabs for taking baits was the combined take by monitor lizards, toads, unknown, and ants, the latter being omnipresent on both Tarague and Orote.

Including the 7 materials tested in the present study, a total of 15 materials have been evaluated since 2001 as parachute/flotation materials for broadcast application of baits for brown treesnakes (Shivik et al. 2002, Savarie and Tope 2004). With an overall canopy entanglement of only 21% (8/39) the jute netting is the worst of the 15 materials. The 2 biodegradable parachutes, Ecofilm(R) and paper towel, and the 4 commercial paper products (single and double white flags, paper cup, paper plate) were good for hanging-up in the canopy. However, a problem with the 2 parachutes and the paper plate and paper cup is that threads have to be secured to them for attaching the dead mice. This is a time consuming effort and with authorized broadcast application rates of 36 baits/hr (3,600 baits/100 hr), resources are not adequate to prepare the thousands of baits needed for large-scale control of snakes.

From an operational management perspective, the single and double white flags, also called streamers, appear to have the best potential as materials for delivering baits to the canopy. They are commercially available and dead mouse baits can be glued to them without the tedious effort of attaching threads. In addition to hand broadcast, they can be deployed by an electro-mechanical dispenser from helicopters and fixed-winged aircraft. An automatic dispenser is not available for any of the other types of flotation materials. The flags are also biodegradable, however, a negative aspect of this is that water (rain) destroys the paper streamer within seconds, allowing the cardboard with attached bait to fall to the ground (P. Savarie, personal observation). Additional testing with a more water resistant paper will be conducted. streamers, even with the inherent advantages of the paper automated systems for treating dead mice with acetaminophen and for attaching the dead mouse to the cardboard are needed to help make the aerial baiting system as efficient as possible.

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## LITERATURE CITED

BTSCC (The Brown Tree Snake Control Committee). 1996. The Brown Tree Snake Control Plan. U.S. Fish and Wildlife Service. Aquatic Nuisance Species Task Force. Honolulu, Hawaii, USA.

DRAHOS, N. 2002. The mysteries and histories of Guam birds. N. Drahos, publisher. Aurora, New York, USA.

ENGEMANN, R. M., and D. S. VICE. 2001. Objectives and integrated approaches for the control of brown tree snakes. Integrated Pest Management Review 6:59-76.

FRITTS, T. H. 1988. The brown tree snake, *Boiga irregularis*, a threat to Pacific Islands. U.S. Fish and Wildlife Service. Biological Report 88(31). Washington, D.C., USA.

FRITTS, T. H., N. J. SCOTT, Jr., and J. A. SAVIDGE. 1987. Activity of the arboreal brown tree snake *(Boiga irregularis)* on Guam as determined by electrical outages. The Snake. 19:51-58.

FRITTS, T. H., and M. J. McCOID. 1991. Predation by the brown tree snake on poultry and other domesticated animals in Guam. The Snake 23:75-80.

FRITTS, T. H., M. J. McCOID, and R. L. HADDOCK. 1994. Symptoms and circumstances associated with bites by the Brown Tree Snake (Colubridae: *Boiga irregularis*) on Guam. Journal of Herpetology 28:27-33.

FRITTS, T. H., M. J. McCOID, and D. M. GOMEZ. 1999. Dispersal of snakes to extralimital islands: incidents of the Brown Treesnake, *Boiga irregularis*, dispersing to islands in ships and aircraft. Pages 209-223 in G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

HALL, T. C. 1996. Operational control of the brown treesnake on Guam. Proceedings of the Vertebrate Pest Conference 17:234-240.

JOJOLA-ELVERUM, S. M., J. A. SHIVIK, and L. CLARK. 2001. Importance of bacterial decomposition and carrion substrate to foraging brown treesnakes. Journal of Chemical Ecology 27:1315-1331.

RODDA, G. H., and T. H. FRITTS. 1992. The impact of the introduction of the colubrid snake *Boiga irregularis* on Guam's lizards. Journal of Herpetology 26:166-174.

RODDA, G. H., T. H. FRITTS, and P. J. CONRY. 1992. Origin and population growth of the Brown Tree Snake, *Boiga irregualris,* on Guam. Pacific Science 46:46-57.

RODDA, G. H., M. J. McCOID, T. H. FRITTS, and E.W. CAMPBELL III. 1999. Population trends and limiting factors in *Boiga irregularis*. Pages 268-284 *in* G. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

SAVARIE, P. J., J. A. SHIVIK, G. C. WHITE, J. C. HURLEY, and L. CLARK. 2001. Use of acetaminophen for large-scale control of brown treesnakes. Journal of Wildlife Management 65:356-365.

SAVARIE, P. J., and K. L. TOPE. 2004. Potential flotation devices for aerial delivery of baits to brown treesnakes. Proceedings of the Vertebrate Pest Conference 21:27-30.

SAVIDGE, J. A. 1987. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660-668.

SHIVIK, J. A., P. J. SAVARIE, and L. CLARK. 2002. Aerial delivery of baits to brown treesnakes. Wildlife Society Bulletin 30:1062-1067.

USFWS (U.S. Fish and Wildlife Service). 2001. Assessment of the coconut crab *(Birgus latro)* population on the Naval Computer and Telecomunications Master Station. U.S. Fish and Wildlife Service. Guam National Wildlife Refuge. Dededo, Guam, USA.

VICE, D. A., and D. L. VICE. 2004. Characteristics of brown treesnakes *Boiga irregularis* removed from Guam's transportation network. Pacific Conservation Biology 10:216-220.

WILES, G. J. 1987. Current research and future management of Marianas fruit bats (Chiroptera: Pteropodidae) on Guam. Australian Mammalogy 10:93-95.

ZAR, J. H. 1984. Biostatistical Analysis. Prentice Hall. Englewood Cliffs, New Jersey, USA.

# AN EVALUATION OF PASSIVE THERMAL FUMIGATION FOR BROWN TREESNAKE CONTROL IN SURFACE TRANSPORTATION FROM GUAM

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***Abstract:*** The brown treesnake (BTS) has been on Guam for about 50 years and in this period has caused extensive ecological, economic, and social damage. It has also repeatedly dispersed from Guam via the transportation network, arriving at numerous locations. However, the conditions snakes face in surface shipping are unknown, making assessment of the risk of snake survival impossible. To address this, we recorded thermal conditions in surface shipments leaving Guam and identified factors that determine these conditions. We monitored 16 shipments to locations in Micronesia and the United States mainland and conducted a series of intensive studies at the Naval dock facility, using up to 29 containers at a time. Maximum temperatures recorded while containers were in transit were likely too low to consistently kill snakes. Empty exhibited uniformly high temperatures, but filled containers did not heat as much, nor as evenly. Maximum temperatures inside boxes and furniture are even lower, though often still exceed 40° C. Exposed containers reached high temperatures, but shading by other containers greatly decreased the maximum temperature reached inside.

***Key Words:*** *Boiga irregularis,* brown treesnake, dispersal, Guam, invasive species, passive thermal, fumigation, surface shipping.

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## INTRODUCTION

The brown treesnake (BTS, *Boiga irregularis*) arrived on Guam in the late 1940s as a stowaway in military cargo. By the early 1980s the snake inhabited the entire island and caused one of the most extensive ecological upheavals due to a single introduced species (Fritts and Rodda 1998, Perry and Morton 1999). Other problems are economic, primarily resulting from frequent and costly snake-caused power outages, and sociological, resulting from human bites and indirect cultural impacts (Rodda et al. 1997, Fritts and Rodda 1998). As dire as this problem has become on Guam, the risk of its further spread is even more alarming. Brown treesnakes have been found on ships and aircraft leaving Guam and at many destinations served from the island, some as far away as Spain (Fritts et al. 1999). Sanitizing the transportation network has repeatedly been identified as a high priority (Brown Tree Snake Control Committee 1996). Despite a substantial annual budget, the agencies conducting interdiction efforts on Guam and elsewhere lack the funding, authority, outside cooperation, and tools to achieve total success. As a result, snakes still leave Guam, although at a reduced rate, via the extensive commercial and military transportation network (Vice et al. 2004).

Currently, the primary control tools used for snake interdiction are traps, night-time visual searches, and dog searches (Vice and Pitzler 2002). However, these methods are expensive, time-consuming, and not completely effective (Vice and Vice 2004). This has led to attempts to develop additional tools for this task, including various methods for cargo fumigation using chemicals (Savarie and Bruggers 1999). A second type of fumigation, thermal fumigation, involves exposing high-risk cargo to thermal conditions that would lead to snake mortality (Christy et al. 2007). Since they lack the ability to physiologically control their body temperature, exposure of reptiles to either cold or heat can be fatal. Operationally, this means that temperatures in airplane wheel wells might drop low enough during a flight that snakes would freeze to death (Perry 2002). Thermal fumigation can be passive, relying on available conditions and not requiring additional technology, or active, depending upon special equipment for forced delivery (Zeichner et al. 1998). Thermal fumigation, and especially passive thermal fumigation, has several

advantages over chemical fumigation. Among them are lack of human health hazards, lower cost, and greater environmental acceptability. However, conditions in containers leaving Guam have not previously been assessed, and the utility of thermal fumigation for BTS control is unknown. Here we address this gap through both experimental manipulations of containers waiting to be loaded and observation of conditions in containers being shipped to multiple off-island locations. In conjunction with data on BTS thermal biology (Christy et al. 2007), we then assess the utility of thermal fumigation for BTS control in the Guam transportation network.

## METHODS

### Guam's Transportation Network

We obtained information on the procedures containers are exposed to in both civilian and military operations from written material (Vice et al. 2004), the shipping companies, port operations personnel, USDA Wildlife Services specialists tasked with inspecting cargo, and site visits. In addition to understanding standard procedures, we looked for opportunities to expose occupied shipping containers to intense solar radiation while minimizing the negative impact on shipping operations.

### Conditions In Containers

We used three approaches to document temperatures inside containers. First, we measured temperatures in shipments made between 1996 and 1999. In each shipment, we installed temperature-sensitive dataloggers (Hobo loggers, Onset Computer, Pocassett, Massachusetts) set to sample at 5-20 minute intervals, depending on the study. Our study encompassed goods traveling to the United States (US) mainland (14 dataloggers retrieved from 7 shipments) and all three major islands in the Commonwealth of the Northern Mariana Islands (CNMI): Rota (8 dataloggers retrieved from two shipments), Saipan (6 dataloggers retrieved from three shipments), and Tinian (5 dataloggers retrieved from 3 shipments). Cargo utilized varied to the extent possible, and included both commercial shipments and household goods. This procedure provided information on conditions in transit, which typically took a month or more. However, in most cases we had to rely on the good will of shippers for both placement and retrieval. This resulted in a small sample size and allowed little control of logger placement within a container and of container placement on the ship. Additionally, it allowed for little replication within a shipment. Given the limitations of the initial study, we also conducted studies of conditions in shipping containers in route to the CNMI. In one container in route to Saipan and one heading to Tinian, we were able to precisely place a large number of loggers inside the cargo and document the placement of the container on the ship.

To experimentally examine the effects of specific operational procedures on temperatures inside containers, we conducted studies simulating dock-side conditions at the US Navy yard on Guam. In single-container studies we used both empty and full 20-ft containers placed on the dock. Initially, we only used an empty container in which we placed eighteen dataloggers, set to record every 5 minutes before it was sealed for 3 days. Although most containers leaving Guam are empty, the greatest risk of snake incursion is likely posed by containers packed on Guam. We therefore also simulated the dockside leg of the process of shipping containerized goods. We used two 20-ft containers, one containing goods and one, serving as a control, that was left empty. In the first phase of the study, we simulated shipping of a vehicle by enclosing a van in the container for 10 days. A second set of dataloggers was installed within and about the vehicle. In the second phase, we removed the van and filled the cargo container with wooden shipping pallets, also containing dataloggers.

In operational use, containers are typically stacked, usually two-high, in order to save space. Their spatial arrangement tends to be aggregated, in dense rows. As a result, some containers are exposed to sunlight from above and all four sides, others get sun only from above and perhaps from their ends, and some get little or no direct sunlight. In a final container study we used 29 empty 20-ft shipping containers and a design which allowed us to sample containers differing in contents, dock tenure, and location within the cargo yard. The containers were arrayed in three rows, some of them stacked. We placed a single logger, set to sample every 5 minutes, in each of 17 accessible containers. All loggers were placed in the same relative position within the container: on the floor, in the middle, 3 ft from the door. Containers were then sealed for one month.

## RESULTS

### Guam's Transportation Network

Much of the cargo shipped from Guam is first handled in a multitude of privately owned and operated warehouses located in the Harmon industrial area. Empty containers are stored nearby or delivered as needed, often having been placed in high snake density areas for protracted periods. Some outbound material is kept outdoors for prolonged periods, usually in an un-containerized state highly susceptible to snake incursion. Normally, however, outbound materials are kept indoors (exceptions include private vehicles, a large number of which are shipped off island every year). Typically, they are only placed in reliably snake-resistant containers shortly before they are taken to the port. The normal procedure used at these facilities, therefore, does not provide an opportunity for lengthy exposure to sunlight. Further, most packing installations lack the space for storing sealed containers outside for any length of time. In contrast, containerized and breakbulk (non-standard goods, such as pipes, that cannot fit inside a container) cargo on Guam often spend several days at the port. Although some containers are stuffed at the last minute and loaded directly from the truck onto the ship, cargo (including all containers and most breakbulk) is typically stored uncovered at the port. This occurs while it is waiting to be loaded onto the ship, or after unloading and before pickup by the consignee. During sunny periods, these containers are often exposed to strong sunlight, but snakes are free to enter or leave breakbulk throughout this period.

After staging, containers are stacked onto ships. At this stage, there is a large difference between some of the small barges, which typically serve short-range destinations such as the CNMI, and the larger ships serving more distant locations. Barges typically store containers in a single layer, all on deck, whereas larger ships store containers in multiple layers, many of them under the deck and completely isolated from sunlight. Thus, the dock stage appears to be the most promising and least disruptive part of the process for the purpose of passive thermal fumigation.

### Conditions In Containers

About ten percent of the loggers installed in shipments were not recovered. A typical profile obtained from a shipment of household goods from Andersen Air Force Base to the mainland US can be divided into 6 periods (Figure 1). Initially (A), goods and loggers are inside the temperature-controlled house. They are then crated, and for a period (B) await loading on the dock, inside metal containers. This is when temperatures reach their highest levels. The container is then loaded onto a large ship, typically (but not always) below deck, and remains there throughout its travel to its destination (C). Temperatures can be quite low during parts of this phase. Upon arrival, the goods are transferred to storage (D), where they await delivery to the owner. Finally, they are unpacked inside a temperature-controlled-house (E). The final section (F) represents temperatures recorded as the logger was being shipped back to Guam. Overall, recorded temperatures were mostly moderate. High temperatures were recorded, if at all, when the cargo was on the dock. Of all crossings documented (N = 33 loggers in 15 shipments), the highest temperature recorded was 52.9° C and the lowest maximal temperature recorded was 29.8 °C (mean maximum temperature and SD = 40.0 $\pm$ 6.99). The lowest temperature recorded was −12.8° C and the highest minimum temperature seen was 29.1 °C (Mean and SD: 15.3 ± 13.26).

A somewhat different pattern was found in cargo shipped to the CNMI, although the distinction between temperatures recorded on the dock and on the ship was retained. Temperatures measured in the pallet kept within the warehouse until shipping (Figure 2A) were considerably lower than those seen in the pallet left outside (Figure 2B). However, both pallets reached similar temperatures during the latter part of the process, including time on the dock on Guam, in transit, and on the dock on Rota. This is because shipments to the CNMI were carried by open barges which typically stored containers in a single layer on an open deck.

Shipments to the mainland experienced significantly lower maximum (Mainland: 36.7 ± 6.39 °C, Saipan: 47.8 ± 2.78 °C, Rota: 45.5 ± 8.17 °C, Tinian: 39.1 ± 2.25 °C; $F_{1,7.95} = 8.7$, $p = 0.0185$) and minimum (Mainland: 1.8 ± 7.70 °C, Saipan: 25.2 ± 0.88 °C, Rota: 26.2 ± 1.74 °C, Tinian: 24.6 ± 2.06 °C; $F_{1,9.77} = 30.87$, $p = 0.0003$) temperatures. The average high temperature measured for all shipments to the CNMI was 42.4 °C (range: 31.1-52.9) and the average low temperature was 25.6 °C (range: 22.1-29.1).

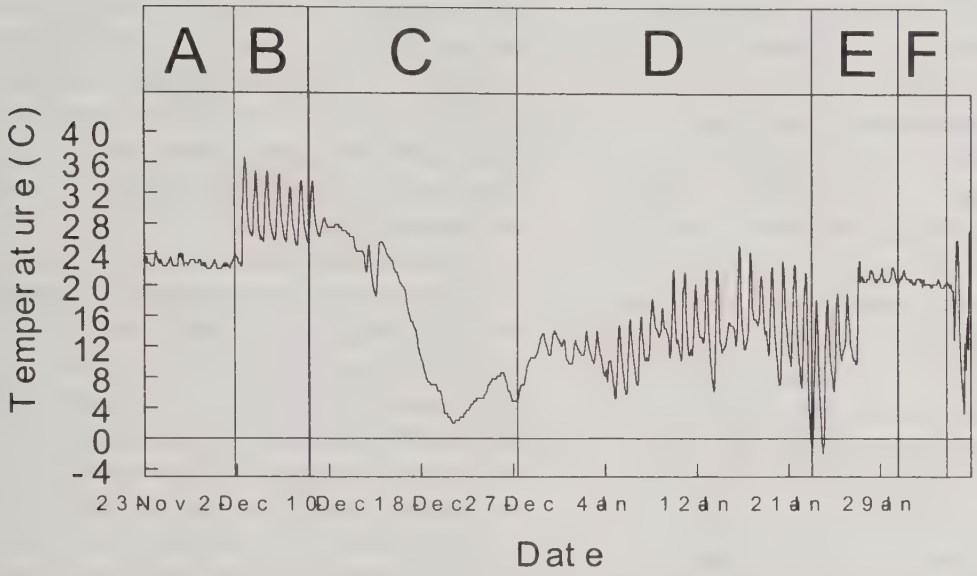



**Figure 1.** Temperatures recorded inside household goods shipped from Guam to the mainland US. A: goods are inside the home. B: crated and containerized goods await loading on the dock. C: the container is on board a large ship. D: following arrival at the destination, goods are in storage. E: goods are inside the destination house. F: temperatures recorded as the logger was being shipped back to Guam.

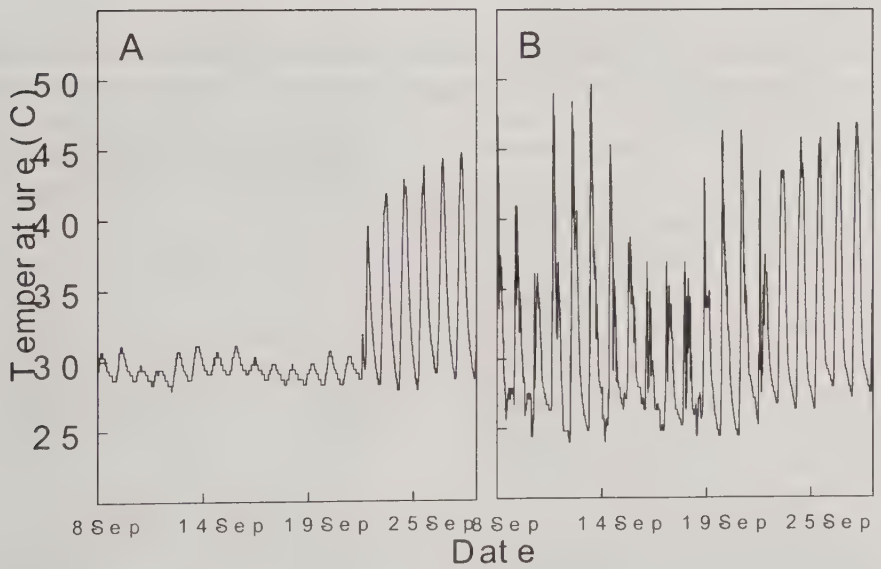



**Figure 2.** Temperatures recorded inside two pallets of fencing materials shipped from Guam to Rota, Commonwealth of the Northern Mariana Islands: (A) a pallet was warehoused until ready to be loaded onto the ship, (B) a pallet left exposed to ambient conditions outside of the warehouse.

We did not detect side or depth effects on mean or maximum temperature measured within a container shipped to Saipan, but logger height had a highly significant effect (Mean: $F_{2,12} = 20.06$, $p < 0.001$; Maximum: $F_{2,12} = 4.06$, $p = 0.045$). We also identified significant differences in maximum temperatures measured inside and outside boxes, as well as those placed within a shipped van ($F_{2,27} = 9.09$, $p = 0.001$). The highest values were measured in the container at large (Mean and SD = 51.5 ± 7.30 °C, range = 39.7-62.7 °C, n = 16), the lowest were inside the van (Mean and SD = 41.1 ± 3.20 °C, range = 35.3-44.9 °C, n = 8), and intermediate values were obtained in boxes placed outside of the van (Meanand SD = 44.5 ± 3.80 °C, range = 40.1-49.0 °C, n = 6). Qualitatively similar results were obtained from the Tinian shipment, with temperatures inside cargo (Mean and SD = 41.3 ± 2.65 °C, range = 38.3-45.4 °C, n = 10) significantly lower than those in the container at large (Mean and SD = 46.0 ± 3.64 °C, range = 40.1-52.4 °C, n = 14; $F_{1,16} = 10.50$, $p = 0.005$). All other single factor effects were also significant ($p < 0.031$ in all cases), with the highest values obtained at the top portion of the middle section of the container.

Mean temperatures recorded within the empty container were near 32 °C, and maximum values ranged from 44.4-50.1 °C (Table 1). Figure 3 presents typical temperature profiles for a variety of locations inside the container. The temperatures measured are summarized below according to logger location. As expected from the similarity of profiles, there was no significant effects due to logger position within the container (for height, $F_{2,11} = 1.68$, $p = 0.230$; for container side, $F_{2,11} = 0.25$, $p = 0.786$; for container end, $F_{2,11} = 2.61$, $p = 0.118$). This indicates no significant stratification occurs within empty containers, with similar temperatures prevailing throughout.

Temperatures were highest in the empty container and lowest inside the van enclosed in the filled container (Figure 4, Table 2). The difference between the two containers was significant ($F_{1,7} = 31.97$, $p = 0.002$), as was the difference between temperatures measured inside the van and in the rest of the experimental container ($t = 2.32$, $df = 16$, $p = 0.034$). Thus, an empty container will heat more, and show less thermal stratification, than a container holding even some goods. Location within the container exerted a significant effect on maximum temperature measured. Temperatures were higher as elevation within the container increased ($F_{2,7} = 92.27$, $p < 0.0001$), were higher at the middle of each container ($F_{2,7} = 7.86$, $p = 0.004$), and were highest on the side most exposed to direct sunlight ($F_{2,7} = 7.98$, $p = 0.004$).

**Table 1.** Summary of maximum daily temperatures obtained inside a single empty 20-ft container. Min is the lowest maximum temperature recorded; Max the highest. Sample size is the number of loggers used.

| | | | Max temperature (°C) | |
|---|---|---|---|---|
| **Hobo Location** | **Sample Size** | **Mean (SD)** | **Min** | **Max** |
| Height | | | | |
| Top | 6 | 47.7 (2.86) | 44.9 | 52.9 |
| Middle | 5 | 47.4 (2.38) | 44.9 | 50.1 |
| Bottom | 7 | 45.6 (1.44) | 44.4 | 48.5 |
| Length | | | | |
| Front | 5 | 48.0 (0.98) | 46.4 | 49.0 |
| Center | 8 | 45.5 (1.27) | 44.4 | 48.0 |
| Back | 5 | 47.6 (3.67) | 44.9 | 52.9 |
| Width | | | | |
| Left | 7 | 46.3 (1.78) | 44.4 | 48.5 |
| Middle | 4 | 47.4 (2.59) | 44.9 | 50.1 |
| Right | 7 | 47.0 (2.88) | 44.4 | 52.9 |

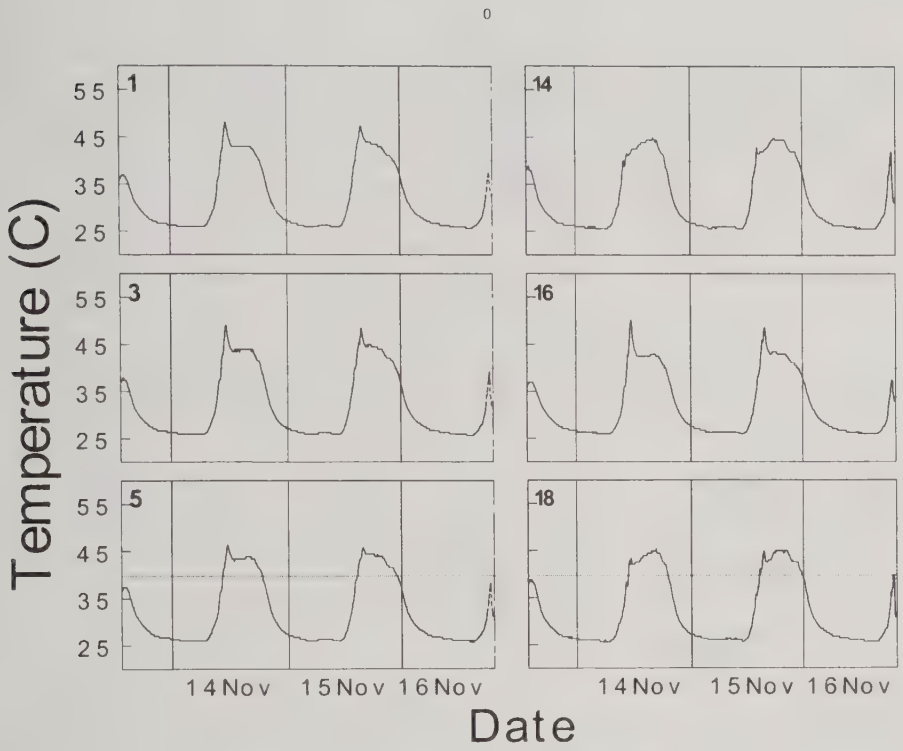



**Figure 3.** Temperatures recorded inside an empty 20-ft container. The dotted line represents 40 °C, presumed harmful to snakes.

We found a similar pattern when the van was replaced with pallets, which do not hold enclosed spaces and allow more air circulation. However, although maximum temperatures were highest in the empty container and lowest inside the pallets, the difference was small and not statistically significant ($F_{1,17} = 1.64$, $p = 0.248$). The difference between temperatures measured in the pile of pallets and in the rest of the experimental container were also not significantly different ($t = 1.14$, $df = 16$, $p = 0.269$). Similarly, although we detected a significant height effect ($F_{2,17} = 5.59$, $p = 0.043$), we found no side effect ($F_{2,17} = 0.31$, $p = 0.746$).

Temperature profiles obtained in multi-container studies were similar in shape to those obtained before (Figure 5). However, overall values obtained were lower than in single-container studies. Mean temperatures were about 30 °C (range: 28.1-30.7). The highest maximum hourly temperature measured was 43.9 °C, and the lowest maximum temperature we measured was 30.7 °C. Containers placed on the top row reached significantly warmer temperatures than those in lower ones ($F_{1,8} = 200.84$, $p < 0.001$, Table 3). However, temperatures typically exceeded 40 °C for less than one hour at a time. Thus, although containers placed on top do reach considerably higher temperatures, this only happens for relatively short periods of time. Repeating the same analysis with data that better represent overall patterns (Table 3) showed no significant effects due to stacking height and location within row (for height $F_{1,8} = 0.02$, $p = 0.891$; for location within the row $F_{1,8} = 0.61$, $p = 0.459$). However, internal rows showed higher temperatures than those measured in external row containers ($F_{1,8} = 10.43$, $p = 0.012$), perhaps because of reflected radiation from adjacent containers.

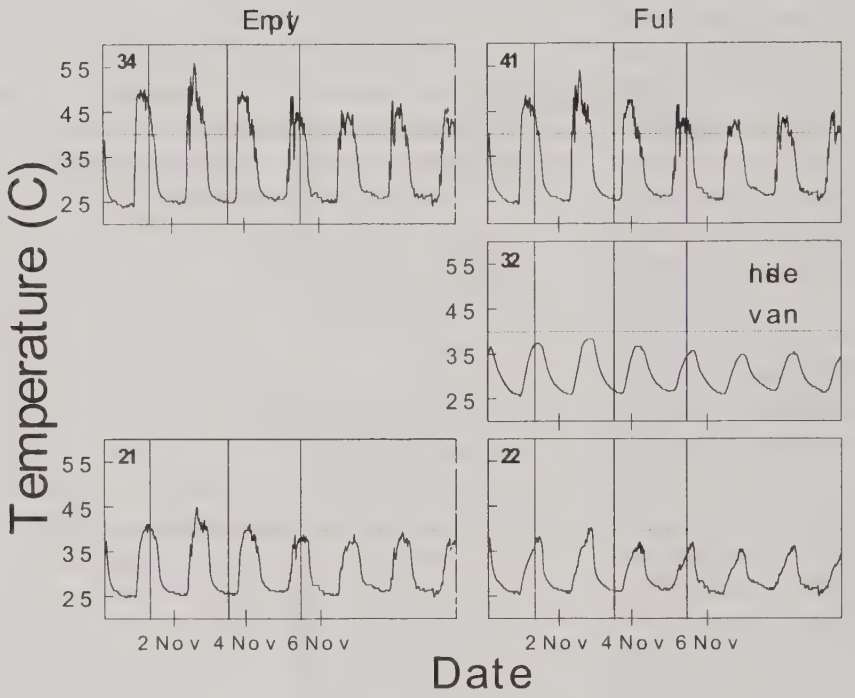



**Figure 4.** Temperatures recorded inside an empty 20-ft container (left) and one containing an experimental load. The dotted line represents 40 °C, presumed harmful to snakes.

**Table 2.** Temperatures measured inside control (empty) and experimental (containing van) 20-ft containers and inside the vehicle enclosed within the experimental container. The first four columns present container mean temperatures, whereas the next two provide the highest (Max) and lowest (Min) temperatures. N is the number of dataloggers used.

| | Temperature (means, °C) | | | Maximum temperatures (°C) | | |
|---|---|---|---|---|---|---|
| Location | Mean (SD) | Min | Max | Lowest | Highest | N |
| Control | 31.1 (0.77) | 24.2 | 50.3 | 43.4 | 56.0 | 13 |
| Experimental | 30.9 (1.04) | 24.4 | 47.9 | 37.9 | 52.5 | 13 |
| Van | 30.9 (0.93) | 25.6 | 41.6 | 37.4 | 49.6 | 5 |

## DISCUSSION

Maximal temperatures reached inside containers leaving Guam were highly variable, ranging from relatively low (about 35 °C) to very high (near 60 °C). Given that shippers and cargo-handlers alike wish to minimize the time they must be responsible for cargo, the optimal timing for passive thermal fumigation to occur would be during periods when the cargo is on board a ship and en route to its destination. Unfortunately, our data conclusively show that this is not a viable option. Typical temperatures during the shipping process were well below the levels that might be useful for snake control. One consistent pattern, however, was that temperatures in containers bound for the CNMI were higher than those measured in US mainland-bound cargo,

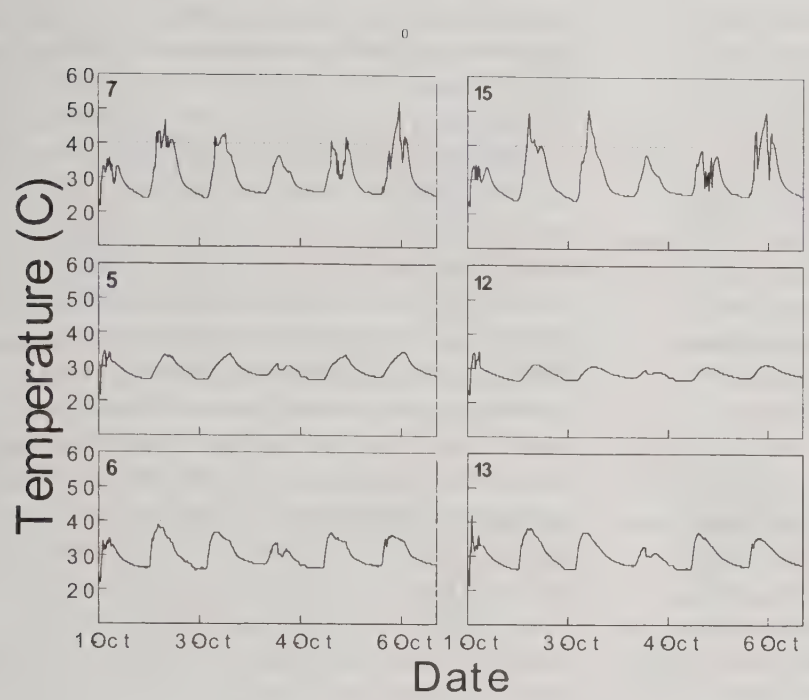



**Figure 5.** Temperatures recorded inside 20-ft containers stacked on the dock on Guam. The dotted lines represents 40 °C, presumed harmful to snakes.

**Table 3.** Summary of maximum daily temperatures obtained in the multi-container study. N is the number of containers in each category that were sampled. Min is the lowest maximum temperature recorded; Max the highest.

| | | Maximum temperature (°C) | | | | | |
|---|---|---|---|---|---|---|---|
| | | Hourly | | | Overall | | |
| Location | N | Mean (SD) | Min | Max | Mean (SD) | Min | Max |
| Stacking height | | | | | | | |
| Top | 5 | 37.5 (5.60) | 31.1 | 43.9 | 62.5 (2.06) | 59.2 | 64.2 |
| Bottom | 11 | 36.5 (4.46) | 30.7 | 42.9 | 36.5 (3.73) | 31.9 | 43.9 |
| Row | | | | | | | |
| External | 10 | 34.8 (3.84) | 30.7 | 41.1 | 44.4 (12.61) | 33.2 | 64.2 |
| Internal | 6 | 40.2 (4.12) | 33.2 | 43.9 | 45.0 (14.44) | 31.9 | 64.2 |
| Location within row | | | | | | | |
| Corner | 6 | 38.5 (5.95) | 30.7 | 43.9 | 46.5 (11.53) | 33.2 | 62.0 |
| Internal | 10 | 35.9 (3.75) | 31.4 | 42.5 | 43.5 (14.04) | 31.9 | 64.2 |

presumably because the latter are stowed below-deck and the former are often above-deck and exposed to additional solar radiation. When the cargo is waiting on the dock, maximum temperatures occasionally exceeded 60 °C, and often surpassed 40 °C, a temperature identified by Christy et al. (2007) as lethal to BTS after moderate exposure. Thus, if passive thermal fumigation is to be useful, it will have to occur during the dock phase of the process.

About 75 percent of the containers leaving Guam are empty. Temperatures inside empty containers are typically high, and may effectively kill or harm snakes trapped inside. This suggests

that air movement inside the containers is considerable and allows temperatures to equalize throughout. However, empty containers in good condition (i.e., lacking holes) pose a relatively low risk of snake incursion when their doors are closed. Stuffed containers are more risky, both because they have to be kept open during the loading process and because the cargo may have been invaded by a snake before being loaded. Full containers consistently showed a significantly different thermal profile than empty ones. They did not heat to the same extent, and loggers placed inside boxes and other cargo reached even lower maximum temperatures. Tightly-packed containers will allow less air circulation than ones left partly empty. As a result, containers packed full of boxes will provide more hospitable hiding places for snakes than ones that are half-packed with contents allowing easy air circulation, as in our pallet study. A snake loose within a container will be free to seek the least stressful environment it can find. Unfortunately, temperatures inside containers full of actual cargo were significantly stratified. The maximal temperatures measured near the bottom of the container or inside cargo did not typically reach 40 °C, possibly providing a refuge which could allow a snake to survive through the relatively brief periods of high ambient temperature elsewhere in the container.

When sunlight is abundant, maximal temperatures are reached inside sealed containers quite rapidly. This suggests that, when ambient conditions are right, only a relatively short period would be needed for passive thermal fumigation to reach its maximum possible efficacy. Conveniently, containers often stay on the dock in Guam for at least two days before being loaded, and two full sunny days should be considered the minimum amount of time allotted for this process. Importantly, four half-sunny days are not equivalent to two sunny days. Containers protected from direct sunlight do not typically reach high temperatures, and even those exposed to direct sunlight do not exceed 40 °C for very long. Thus, to receive any benefits from thermal fumigation, containers should not be stacked. Moreover, spacing between them must be great enough to minimize shading and maximize exposure to direct sun. During particularly cloudy periods, which are especially common during the rainy season, this may not be logistically possible, given shipping schedules, and alternative methods would then have to be applied.

Under the right conditions, passive thermal fumigation offers an economically attractive tool for limiting BTS excursion from Guam. Moreover, receiving destinations in the Pacific (e.g., CNMI and the Federated States of Micronesia) can use passive thermal fumigation, in combination with barrier technology (Perry et al. 1998, 2004; Rodda et al. 1998), to improve the chances that any snakes arriving in cargo will be dead before the container is opened. However, it is important to recognize the limitations of this method. Although likely to sometimes be effective, passive thermal fumigation is not a reliable tool unless used conscientiously, consistently, and intelligently. Sunny days cannot always be guaranteed, and container contents and packing will have a major effect on conditions inside. As a result, identical containers placed in the same location on consecutive days might reach greatly different internal temperatures. Similarly, different containers experiencing the same conditions may reach different internal temperatures. This level of complication is unlikely to present an appealing and effective operational solution under conditions prevalent in the Pacific, which unfortunately makes passive thermal fumigation not a "silver bullet" solution. Other methods of sanitizing cargo will be necessary, and passive thermal fumigation will be especially useful if used in conjunction with means for isolating and interdicting cargo, such as snake barriers and active thermal fumigation.

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## LITEARTURE CITED


Brown Tree Snake Control Committee. 1996. The brown treesnake control plan. U.S. Fish and Wildlife Service, Honolulu, Hawaii, USA.

Christy, M., R. Bischof, G. Rodda, and J. Savidge. 2007. Can temperature be used as a tool for limiting brown treesnake invasion via transportation pathways? Pages 246-256 *in* G. W. Witmer, W. C. Pitt, and K. A. Fagerstone, editors. Proceedings of the managing vertebrate invasive species. August 2007. Fort Collins, Colorado, USA.

Fritts, T. H., and G. H, Rodda. 1998. The role of introduced species in the degradation of island ecosystems: a case history of the brown treesnake, *Boiga irregularis*, on Guam. Annual Review of Ecology and Systematics 29:113-140.

Fritts, T. H., M. J. McCoid and D. M. Gomez. 1999. Dispersal of snakes to extralimital islands: incidents of brown treesnake (*Boiga irregularis*) dispersing to islands in ships and aircraft. Pages 209-223 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

Perry, G. 2002. Conditions facing airplane stowaways during flight: temperature in wheel-wells and cargo compartments. Aviation & Space Environmental Medicine 73:673-676.

Perry, G., and J. M. Morton. 1999. Regeneration rates of the woody vegetation of Guam's Northwest Field following major disturbance: land use patterns, feral ungulates, and cascading effects of the brown treesnake. Micronesica 32:125-142.

Perry, G., E. W. Campbell, III, G. H. Rodda, and T. H. Fritts. 1998. Managing island biotas: brown treesnake control using barrier technology. Pages 138-143 *in* R. O. Baker and A. C. Crabb, editors. Proceedings of the 18th Vertebrate Pest Management Conference. University of California, Davis, California, USA.

Perry, G., G. H. Rodda, T. H. Fritts, and F. J. Qualls. 2004. Snake control using barrier technology: a summary of work conducted 1995-2001. Micronesica 37:177.

Rodda, G. H., T. H. Fritts, and D. Chiszar. 1997. The disappearance of Guam's wildlife. BioScience 47:565-574.

Rodda, G. H., T. H. Fritts, G. Perry, and E. W. Campbell, III. 1998. Managing island biotas: can indigenous species be protected from introduced predators such as the brown treesnake? Transactions of the 63rd North American Wildlife and Natural Resources Conference.

Savarie, P. J., and R. L. Bruggers. 1999. Candidate repellent, oral and dermal toxicants, and fumigants for brown treesnake control. Pages 417-422 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

Vice, D. S., M. A. Linnell, and M. E. Pitzler. 2004. Summary of Guam's outbound cargo handling process: preventing the spread of the brown treesnake. Draft, USDA/APHIS/WS report, Barrigada Heights, Guam.

Vice, D. S., and M. E. Pitzler. 2002. Brown treesnake control: economy of scales. Pages 127-131 *in* L. Clark, editor. Human conflicts with wildlife: economic considerations. USDA National Wildlife Research Center, Fort Collins, Colorado, USA.

Vice, D. S., and D. L. Vice. 2004. Characteristics of brown treesnakes removed from Guam's transportation network. Pacific Conservation Biology 10:216-221.

Zeichner, B. C., A. L. Hoch, and D. F. Wood, Jr. 1998. Heat and IPM for cockroach control. The IPM Practitioner 20:1-6.

# SPOTTING CRYPTIC ANIMALS IN THE DARK: WHAT LIGHT PROPERTIES SHOULD A GOOD HEADLAMP HAVE?

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***Abstract:*** Relying on headlamp illumination for visual detection of cryptic nocturnal animals may present a challenge. To test how search light properties affect brown treesnake (*Boiga irregularis*) detection rate, we assigned eight biologists to search for dead snakes placed in roadside vegetation. Each person conducted 4 searches using lamps with varying properties: weak versus strong light, crossed by narrow versus wide beam. On each occasion, 100 snakes were placed randomly along the roadside transect. The mean number spotted per transect search was 13.5. Using an information theoretic approach, sequential order of transect runs was the only confounding variable included in the model with the highest support: 1.5 fewer snakes (95% CI = -0.4 to -2.5) were found for every sequential transect search a person conducted. A narrow beam spotlight rendered almost seven fewer snakes per search than a broad beam floodlight (95% CI = -4.5 to -9.2). A weak light rendered 4.5 fewer snakes than a strong light (95% CI = -2.1 to -6.9). We suspect that the benefit of using a lamp with a floodlight beam is particularly pronounced when a complex, 3-dimensional forested habitat is surveyed and when the traveling speed is relatively high.

***Key Words***: *Boiga irregularis*, brown treesnake, floodlight, headlamp, invasive species, light source, power, search efficacy, spotlight, visual search.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Animal surveys often use visual searches as a detection tool. For nocturnal animals, this may present a challenge. In some taxa, the *tapetum lucidum* layer of the retina reflects light. Using a light source held close to the observer's own eyes, the observer may, therefore, be able to locate a distant animal by its 'eye shine' (Ribi 1981). This is the case for many, though not all, nocturnal animals in a wide range of taxa, including invertebrates, fishes, amphibians, crocodiles, and certain birds and mammals (e.g., Stavenga et al. 1977, Somiya 1980, Bearder et al. 2006). In snakes, however, the eyes normally do not reflect much light (although see Henderson 2002). Therefore, detecting a snake in the dark typically relies on spotting the characteristic shape or the somewhat different sheen of the animal compared to the surrounding habitat.

Most field workers studying nocturnal snakes use headlamps to free their hands for capturing and handling snakes or to take notes. The physical light properties of headlamps used by field herpetologists vary substantially. Many use spotlights with a narrow beam angle, while others use floodlights with a broader beam angle; some use light weight, low power lamps, while others rely on rechargeable cells that allow for more powerful lamps to be used. While many herpetologists have a favored headlamp type (sometimes advocated with great emphasis), rarely are the effects of these headlamp differences on animal detection rates addressed.

We work on a nocturnal, arboreal snake – the brown treesnake (*Boiga irregularis*) – on the Pacific island of Guam. This species (henceforth referred to as BTS) has caused the demise of many of Guam's native vertebrate species (Savidge 1987a, Rodda and Fritts 1992), and now threatens to colonize other Pacific islands (Fritts 1987, Savidge 1987b, McCoid and Stinson 1991, Kraus and Cravalho 2001). Much of our field work occurs at night, and visual searches are an important research and management tool, instrumental in ongoing BTS control efforts on Guam (e.g., Engeman et al. 1998). The multi-agency 'Rapid Response Team,' tasked with traveling throughout the Pacific region to respond to credible BTS sightings, also relies on nocturnal visual searches as

an important tool for snake intervention (alongside trapping and detector dog searches).

To date, the BTS project has used two main types of headlamps. The first, used only on Guam due to restrictions on air transport of wet cell batteries, is a powerful spotlight lamp. The Rapid Response Team also uses a lightweight headlamp that runs on 4 C-size batteries. When searching for snakes, the latter is normally set to the halogen spot mode, rather than the less powerful LED flood mode. Both these headlamps have a very narrow light beam (about 7° and 9°, respectively, when set to spot mode) and can, thus, be characterized as true spotlights. We suspected, however, that it might be beneficial to use a wider floodlight beam when searching for snakes in dense vegetation. We decided to estimate the effect of beam width and light intensity on snake detection rate. To our knowledge, no formal test of this kind has previously been conducted, at least not focusing on amphibian or reptile surveys. We also investigated whether lamp properties interacted with detectability in reference to target perch height and distance from the searcher.

## MATERIALS AND METHODS

### Experimental Design

To test the effects of different headlamp properties on snake detection rate we designed an experiment that allowed us to investigate the separate effects of beam width (spot or flood) and light output (strong or weak). To reduce the confounding effect of different reflector properties we used one single headlamp model (Mila® 3-light Digital; manufactured by Mila Design & Tillverkning AB, Haninge, Sweden; www.mila.se/english/) and manipulated the two light characteristics independently. By using either a 5W or a 20W halogen bulb (i.e., not by dimming the lamp by the electronic circuit this lamp model features) we altered the light intensity. By shielding off the beam with a 220 mm long tube-like extension mounted in front of the reflector, and attaching a 22 mm wide iris at the end of the tube, we created a narrow spot light treatment (about 16°) that differed from the non-manipulated wide flood light state (about 94°). While shielding off the beam caused some drop in light intensity in the center of the beam, this effect was considerably smaller than the difference between the strong and weak power states (Figure 1). The lamp treatments obtained by our manipulations were classified as weak spot, weak flood, strong spot, and strong flood. We anticipated that different lamp types may suit different persons. We, therefore, let eight experienced snake searchers on our staff test all 4lamp types once each, allowing us to model searchers as randomized blocks. Each such test meant walking a 1 km roadside transect, looking for snakes in the scrub forest vegetation along the roadside. Since live snakes come and go, and since we wanted a reasonable spotting rate on which to base the analysis, we chose to ignore any live snakes and had the searchers look for dead snakes that we had placed in the vegetation along the transect. The snakes used were BTS that we obtained (dead and frozen) from the USDA Wildlife Services BTS control program. On the day preceding a transect search, we thawed 100 snakes and arranged them according to a stratified randomized snake placement protocol that was unique for each of the eight transects. Snakes were stratified in placement from the transect line (defined by red paint dots sprayed on leaves at eye-height along the edge of the vegetation lining the road side) to as far into the vegetation as 5 m, and from ground level to as high as 4 m above ground. This 20 $m^2$ cross section area perpendicular to the transect formed 20 'cells' (of 1 $m^2$ each). Five of the 100 snakes were allocated for placement in each cell. The exact location within a distance-by-height cell was chosen based on availability of suitable vegetation for snake attachment (using thin, black cable ties). Snakes allocated to the lowest level could also be placed on the ground.

The order with which the cell allocations appeared along the transect was randomized. If a gap in the vegetation prevented placement of a snake in its assigned cell at a certain transect meter mark, we chose the first available position in a random direction parallel to the transect line (i.e., distance-by-height cell assignments were maintained even if the randomized position along the transect was adjusted). Prior to mounting a snake we measured its snout-vent length (SVL) by stretching it along a tape measure. Once mounted, its mid-body position was measured with a precision of ca 0.1 m relative to the transect line and to the ground.

The searchers were familiar with the experimental design, but had no access to the snake placement protocols, and thus no knowledge on where a snake might be. The same roadside was used for all eight transect searches. The vegetation was a secondary forest with plant genera such as *Hibiscus*, *Leucaena*, *Premna*, *Guamia*, *Triphasia*,

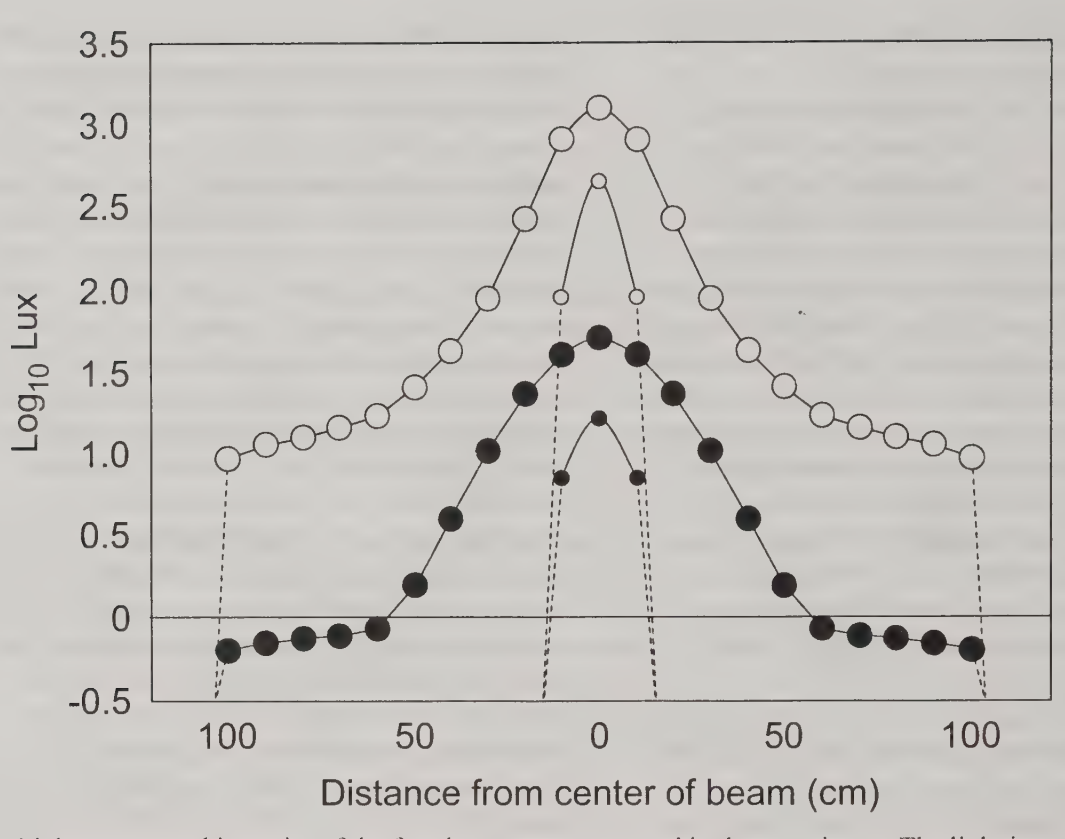



**Figure 1.** Light pattern and intensity of the four lamp treatments used in the experiment. The light intensity was measured with a Lux meter sensor positioned on a flat surface 1 m from the front glass of the lamp. Each symbol is the mean of 8 replicate measures taken in radiating directions from the center of the beam. Filled symbols indicate weak power treatments, empty symbols strong power, large symbols indicate floodlight treatments, and small symbols spotlight treatments. The edges of what the human eye perceives as the distinct beam(s) are indicated by dashed lines.

*Eugenia*, *Morinda*, *Artocarpus*, *Vitex*, *Pandanus*, and various other trees and shrubs.

Due to staff constraints we divided the study in two parts: a first phase in which 4 persons searched on 4 nights (each person conducting a transect search with a different lamp type on each of 4 nights) and a second phase for another 4 persons to conduct their 4 searches. We anticipated that searchers might get increased experience of the search method as the study progressed, so within each group of 4 searchers we let the order with which they tested the different lamps be determined by a randomized Latin square design. The searches took place in darkness (starting >50 minutes after sunset; no searches were scheduled during a full moon).

A searcher was sent out on a transect search accompanied by a note taker who had two roles in addition to data collection. First, the note taker made sure the searcher's walking speed during the search was kept at a constant 0.5 km/hr, a task aided by meter mark signs every 5 m on the transect and data sheets indicating times that certain meter marks should be passed to maintain the correct pace. Brief pauses for taking notes were not part of the 2-hr search. Secondly, the note taker associated each snake spotted by the observer with any nearby snakes' previously measured locations to determine which snake was found. This was possible since the data sheets stated the positions (distance perpendicular to the transect, height, and meter mark) of all snakes. Needless to say, this information was not disclosed to the searcher. The matching was made even easier for the note taker since the data sheet also said whether a snake had a stretched-out pose (85% of the snakes) or if it was coiled (15%). In case two nearby snakes were at obvious risk of being confused with one another, we normally placed one in a stretched position while the other was coiled.

**Measuring Snake Visibility Through Vegetation**

We expected reduced detection at greater viewing distances. This is a universal pattern used, for example, in distance analysis to estimate population densities (Buckland et al. 1993).

However, in our experiment, we did not use the information for density estimation but to discern if such a decline over distance was due primarily to the greater probability of vegetation obscuring a snake with increasing distance or for other reasons (e.g., a snake appearing smaller and less-well lit and therefore harder to see at greater distance). To describe the vegetation effect at different distances from the transect we pooled the vertical layers. Snakes were thus classified and grouped only in horizontal 1-m intervals from the transect line. A searcher walking a transect line does not look only perpendicularly into the vegetation, he/she also looks at some angle ahead, and, occasionally, somewhat backwards. To quantify the value of these oblique viewing angles, we prepared a 14-meter long rope by attaching tags every 1 m, the tags marked from -7 m to +7 m. During daytime we pulled this rope along the transect line and stopped when the 0-m tag was perpendicular to a focal snake. Our surveyor (who had good eye sight) located the snake and memorized its position. He then tried to spot the snake when standing on top of each of the fifteen tags on the rope. If the snake could be spotted from a meter mark tag, it scored as 1 for that situation; if not, 0. For each snake subjected to this procedure, we got 15 such scores. We deemed it unlikely that anyone would spot a snake from a more oblique view point.

We performed this test for 25 snakes on each of the 8 transects. On transect 1 and 5, visibility of snakes #1-25 were scored; on transect 2 and 6, snakes #26-50 were scored, and so on. We then grouped snakes placed in the five 1-m intervals perpendicular to the transect line (the horizontal snake placement aspect) and calculated the mean spotting success (based on 0/1 data) for each meter step along the survey rope. In this way, the detection rate from the points perpendicular to the snakes, and for every meter mark away from those points (up to ±7 m), was expressed as a percentage. For example, focusing on snakes placed 1-2 m from the transect line, we might find that 90% could be spotted from the transect point perpendicular to the snake; 85% could be spotted at points 1 m on either side of it; 78% could be spotted from points ±2 m away, and so on. Because this was merely an aid to better appreciate the influence of vegetative structure, our analyses were limited to graphical and verbal summarization of the results.

**Statistical Analysis**

Our main question was simply "Do the number of snakes spotted depend on any of the lamp variables?" The counts obtained were far enough from the end points of the distribution (0, 100) to allow use of non-transformed data in the analysis.

We assumed that the eight observers would differ in their ability to spot snakes. To control for this confounding effect we included the random effect variable Observer in the model. It soon became obvious, however, that the snake spotting rate also differed substantially between the eight transects. We, therefore, adjusted for the random effect variable Transect in the analysis. Even though our experimental design cancelled out any systematic bias of treatment order (i.e., searcher experience of the set-up) there could still be an order effect that added unexplained variation to the data. Hence, we included the covariate Sequential order modeled as a linear trend. We tested for an interaction effect between Beam and Power while the other variables were modeled without interactions (partially due to lack of replication). The full model can thus be written as:

y = intercept + Transect + Observer +
Sequential order + Power + Beam + Power
x Beam

where y is the predicted number of snakes spotted during a transect search. The analysis treats both Power and Beam as factorial variables. As pointed out above, shielding off the beam caused an undesired drop in central beam intensity. While it was therefore tempting to treat Power as a continuous variable (taking four values), there were two reasons we did not do so. First, the light intensity in the center of the beam was not representative of the entire beam. In the flood light treatments, the majority of the beam had a light intensity much lower than the corresponding spot treatments. Second, it was not evident which measurement scale to use for the light intensity measure, as this is an issue of questionable linearity in light perception. We chose the most easily interpreted analysis method: treating Power as a category variable, taking only two states (strong, weak).

We used an information theoretic approach to evaluate the relative evidence for alternative candidate models (i.e., when successively dropping one or more independent variables). We first tested the full model to see if the confounding variables (Transect, Observer, and Sequential order) and the interaction between Power and Beam had any effect and would be relevant to include in subsequent models. Variables with little or no

explanatory value were dropped before we defined and estimated four plausible models: two models with mixed effects and two with only fixed effects (PROC MIXED, SAS Institute 2003). To obtain correct parameter counts used for calculation of Akaike's Information Criterion (AIC), we used maximum likelihood (ML) as the estimator instead of the default restricted maximum likelihood (REML) [see also Gurka 2006 for a discussion on ML and REML in linear mixed models]. Hand calculation of the $AIC_c$ statistics [the subscript 'c' indicating a small sample adjustment of the AIC value (Burnham and Anderson 2002)] verified that $AIC_c$ values produced using maximum likelihood in PROC MIXED were correct. Random effect variables should not be subjected to statistical tests, and are here merely treated as confounding nuisance variables. Therefore, we do not report on these effect sizes but instead emphasize the lamp-design variables.

## RESULTS

### How Many Snakes were Spotted?

Of the 100 snakes that were mounted along each transect, observers spotted an average of 13.5. While mean number of snakes spotted by each observer during his/her 4 transect searches varied between 12.0 and 16.8, the effect size of the random factor Observer was estimated as zero. Obviously, the variance within observers was greater than between observers, thus the lack of a between-subject effect. Also the interaction between Power and Beam failed to show an effect (95% CI = +2.0 to -6.5). Transect, however, did have some explanatory value in the full model (the 8 transect nights had mean snake sighting values ranging from 10 to 18.5, resulting in a parameter estimate >0), as did the covariate Sequential order (95% CI = -0.2 to -2.8).

We ended up with 4 plausible models with all containing the main effects of Power and Beam, in the 4 possible combinations with (or without) Transect and Sequential order (neither of them modeled with any interactions). When ranking the 4 models by $\Delta AIC_c$ (Table 1), both of the top models contained the covariate Sequential order. The effect was in the opposite direction than expected: the Observers saw, on average, 1.5 fewer snakes for each consecutive transect search (for the top model, 95% CI = -0.4 to -2.5). The intercept of the top model was 22.9 (95% CI = 19.5 to 26.2). To better appreciate the lamp trait effects, setting Sequential order to 2.5 (i.e., after an observer had conducted half of his/her transect searches) in the top model renders an 'experience adjusted' intercept of 19.2. Given the effect coding it means that approximately 19 snakes should be sighted when a strong floodlight is used.

**Table 1.** Summary of Akaike's Information Criterion ($AIC_c$) statistics for modeling snake sighting rates with different lamp types. Variables are coded as follows: T = Transect; O = Observer; S = Sequential order; P = Power; B = Beam; PB = interaction Power × Beam. Transect and Observer were treated as random variables. All variables but Sequential order were coded as factorial. Models with a $\Delta AIC_c$ <2 have considerable support (Burnham & Anderson 2002). *K* is the number of parameters and includes an intercept; $w_i$ is the Akaike weight.

| Models | -2LogLikelihood | *K* | $AIC_c$ | $\Delta AIC_c$ | $w_i$ |
|---|---|---|---|---|---|
| S + P + B | 169.73 | 5 | 182.04 | 0.00 | 0.51 |
| T + S + P + B | 168.52 | 6[1] | 183.88 | 1.84 | 0.20 |
| T + P + B | 172.32 | 5[1] | 184.63 | 2.59 | 0.14 |
| P + B | 176.38 | 4 | 185.86 | 3.82 | 0.08 |
| T + O + S + P + B + PB | 167.46 | 7[1, 2] | 186.13 | 4.09 | 0.07 |

[1] PROC MIXED (SAS Institute 2003) adjusts for the trace of the matrix of random effect variables in linear models and estimates *K* = 1 for random effects with parameter estimates >0.

[2] The $AIC_c$ value of the full model does not include any penalization for factor Observer since the effect size was estimated as zero.

Both factors Beam and Power affected the snake spotting rate. The effect size of factor Beam in the top model was -6.9 (95% CI = -4.5 to -9.2), meaning that a strong spotlight search should result in 12 snake sightings. The effect of Power was somewhat weaker (-4.5; 95% CI = -2.1 to -6.9), translating to 15 snake sightings during a search with a weak floodlight. Combining the main effects, the model prediction for a search with a weak spotlight was 8 snake sightings.

If we consider spotting 19.2 snakes with a strong floodlight as a baseline value, the decline in predicted snake sightings using other lamp types can be given as a percentage for better data transparency. When swapping to a weak floodlight, the search efficacy drops by 23%, while a strong spotlight results in a 36% decline. Choosing a weak spotlight constitutes a 59% drop in snake sightings relative to a search using a strong floodlight.

**Which Snakes were Spotted?**

Snake detection rate fell off rapidly with distance from the transect line (Figure 2). Also, few of the snakes placed on, or close to, the ground were detected (Figure 3). The grand mean positions of snakes spotted across the four lamp treatments and by the 8 observers were nearer to the observers and slightly higher than the mean positions of the target snakes (Figure 4).

Using the data on the 2800 snake spotting opportunities for which we have SVL data (7 Transects × 4 Observers × 100 snakes; SVL data for snakes on one transect were unfortunately lost), the mean detected snake (N = 375) had an SVL of 941 mm (SD = 349 mm) whereas snakes not spotted (N = 2425) averaged 915 mm (SD = 330 mm). However, the snakes used for setting up the different transects varied somewhat in size (SVL means ranging from 868 to 982 mm). The mean

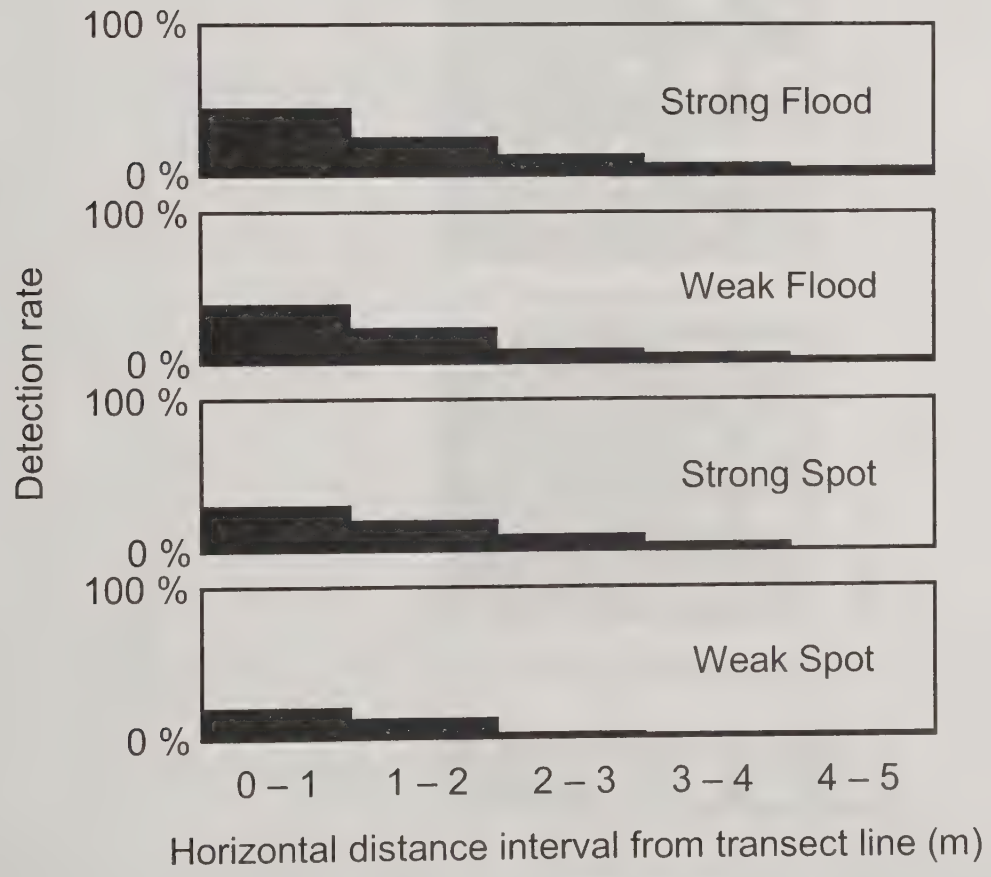



**Figure 2.** Detection rate of BTS by distance from the transect line for the four lamp types tested.

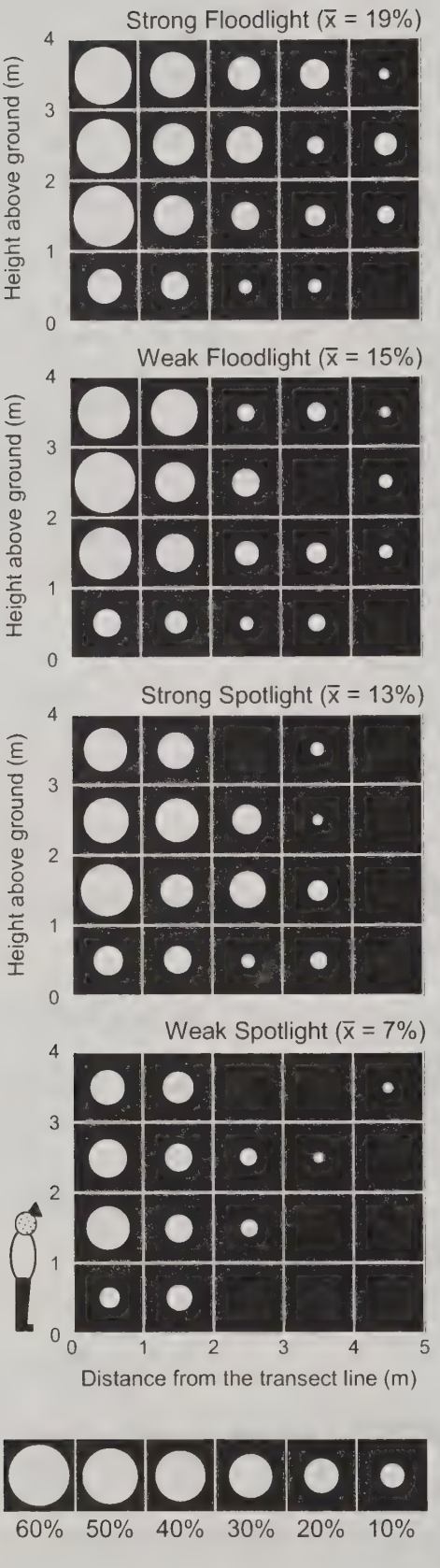



**Figure 3.** Detection rate of BTS per cell for the four lamp types tested. The area of each circular symbol relative to the area of the square 1×1 m cell it sits in correspond to the percentage of BTS spotted in that cell. While the legend below the four panels show 10% increments the size of the symbols are accurate to 1%. Percentages shown are the actual ones; not model predictions.

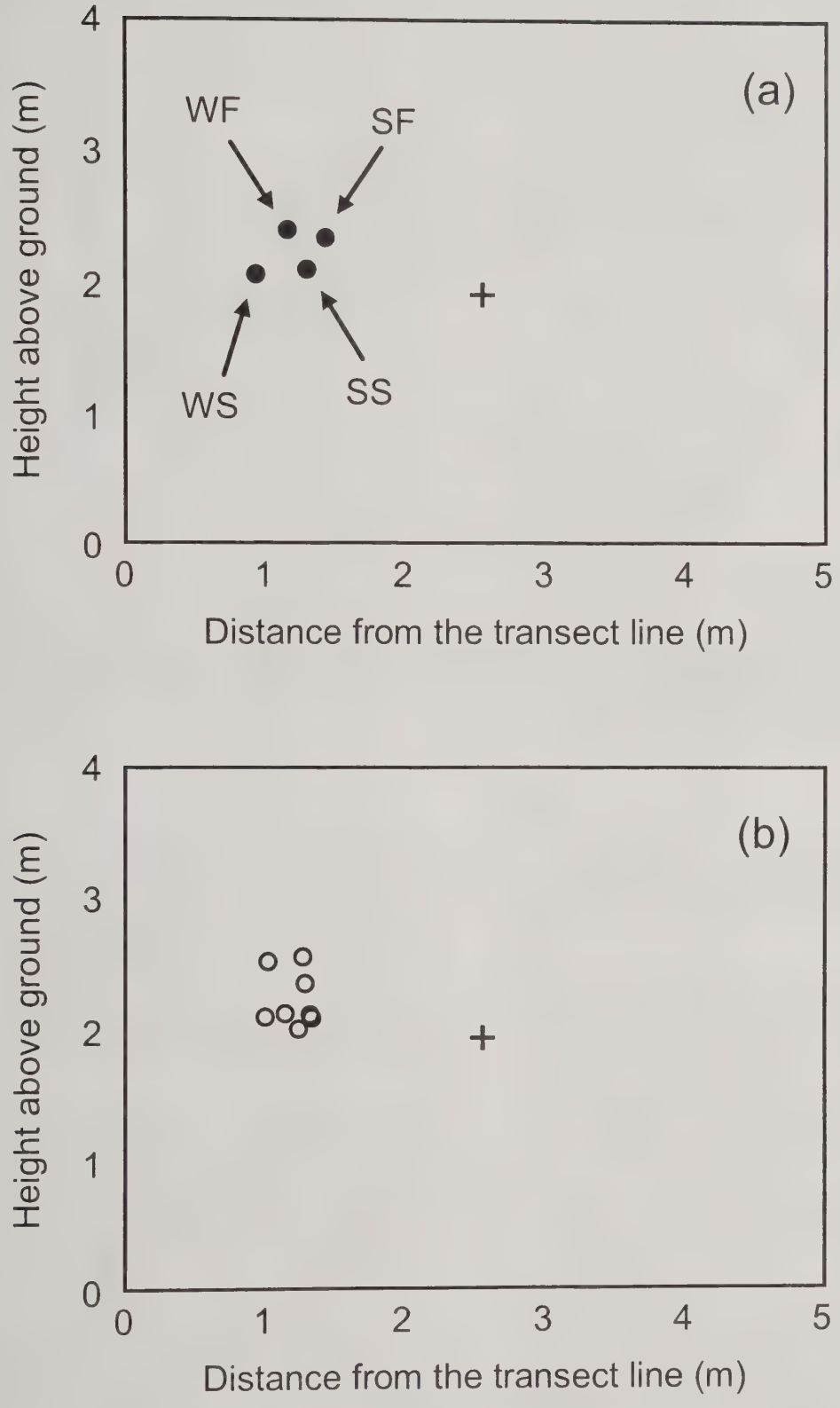



**Figure 4.** Mean positions of (a) BTS spotted with the different lamp types, and (b) by the eight searchers. The + sign indicates the grand mean position of BTS present; SF = Strong Floodlight, WF = Weak Floodlight, SS = Strong Spotlight, and WS = Weak Spotlight.

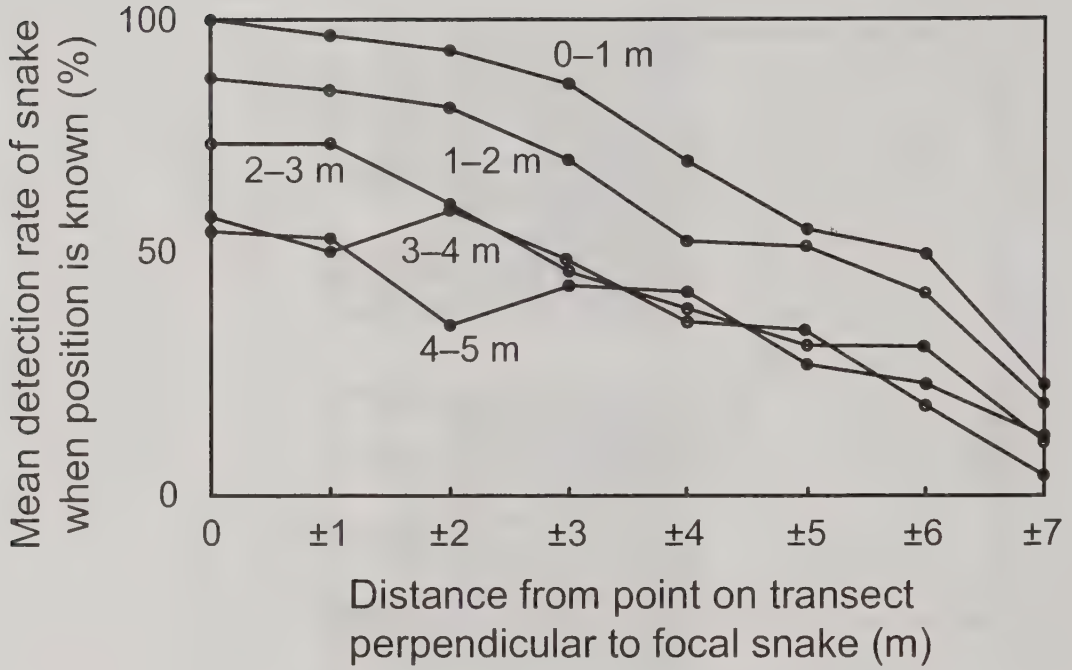



**Figure 5.** The effect of intervening vegetation on BTS visibility: detection rate of BTS known by the surveyor for five meter-wide distance intervals parallel to the transect line (indicated by figures next to curves) as seen from points perpendicular to the focal BTS (x = 0) and progressively more oblique viewing angles up to ±7 m from the perpendicular point. See the text for further explanation.

difference in SVL of snakes spotted versus not spotted on a given transect was 23 mm, with a bias in the direction of larger than average snakes being spotted.

**Measuring Snake Visibility Through Vegetation**

Snakes perched within 1 m of transects had average visibility rates >90% from any sampled point over a distance of 4 m along the transect (i.e., ±2 m from the point perpendicular to a focal snake, Figure 5). The farther one moves away from the point on the transect that is perpendicular to the snake, the longer the distance to the snake and the more vegetation likely to be obscuring it. This is illustrated by the curves in Figure 5 sloping off to the right. While the visibility dropped with a snake's distance from the transect, it must be realized that a lower visibility score – for example, 56% of the snakes 4-5 meters from the transect are visible when viewed from the 0-m mark – does not mean that the complementary figure (in this example 44%) of the snakes equally far into the vegetation cannot be seen from the transect. Even if a snake could not be spotted from one point it might have been visible from some other nearby point. For example, in our sample of 199 snakes (one datum excluded due to uncertainty of the snake's identity and thus its 'true' position) only 5% (N=10) could not be spotted from any of the 15 points we sampled. These "invisible" snakes were situated on average 3.8 m from the transect line and half of them were placed on the ground.

## DISCUSSION

A floodlight beam was important for success in spotting snakes. It appears this effect was stronger than the power aspect; sighting rates dropped only moderately when shifting from a 20W floodlight to a 5W floodlight beam. The latter is good news for field workers that survey remote areas where access to electricity (and the option to re-charge batteries) is limited, since batteries will last four times longer with a 5W bulb than a 20W bulb. What eventually counts, however, is not the wattage as such, but the brightness of the emitted light. Different types of bulbs – and particularly when comparing regular bulbs with diodes – have different energy conversion efficacies, and thus emit more or less light for a given wattage. Given a certain bulb (or diode) type and wattage, there will also be a

reflector dependent trade-off between light intensity and beam width; one may opt for either a brighter spot or a weaker flood illumination. What is the optimal trade-off in this situation may be dependent on several factors, including the habitat structure, how fast the searcher travels, and the study organism itself.

We believe that vegetation structure may have a significant effect on the relative merits of the beam types. When searching less complex vegetation structures – in the extreme, two-dimensional surfaces such as sandy desert, mowed lawns, water surfaces, or fences – a spotlight may be just as good (or better) than a floodlight. For such habitats or structures, vegetation does not obscure the view and a spotlight beam can be held close to the ground (water surface, fence) and the narrow beam used to scan the surface. Since the light is concentrated by a spotlight reflector, the light intensity of the beam, and thus the potential spotting range, increases. This is a good feature in open habitats, but of limited value in densely vegetated habitats. In the latter, a beam that allows a wider peripheral visual field is probably of more importance than the power of the light.

We also anticipate that the speed of walking may have an effect on the relative merits of spotlight versus floodlight lamps. The faster one walks the harder it is to search the entire vegetation of a complex, three-dimensional habitat. That effect is likely to be most evident when a spotlight is used, causing a Beam-by-Pace interaction. Adding the habitat complexity aspect, we may expect a Beam-by-Pace-by-Habitat interaction. It is notable that the headlamp we used (and several similar lamps, all of which feature floodlight reflectors) are designed for night orienteering. This sport is normally conducted in forests where it is important to detect branches, tree stumps, and other obstacles while running as fast as possible between control stations – a situation not dissimilar to searching for animals in a complex habitat under a severe time constraint.

The optimal walking pace will presumably differ not only with the lamp properties and the habitat structure, but also with the focal animals. The harder they are to spot (the smaller or the more cryptic) the slower the walking pace necessary to be to detect a certain proportion of the population within a given distance from the observer. We would thus predict that given a complex habitat, someone surveying small, cryptic animals would benefit more by swapping from a spotlight to a floodlight than would a searcher looking for large animals that stand out from the background.

With the level of replication used, there are issues that we could not resolve. For example, do observers differ in what is a suitable (or not so suitable) headlamp for them personally? This would be manifested as an Observer-by-Lamp interaction. Since each observer tested each lamp type (i.e., each of the four Power by Beam combinations) only once, we cannot test if there is such an effect. It is striking, however, that searchers using the least productive lamp showed a rather remarkable difference in success: the number of snakes spotted varied from 3 to 16 with the weak spotlight. These figures are however confounded by effects of Transect, Observer, and Sequential order with which the lamps were tested, so caution is warranted to not over-emphasize the differences.

**Which Snakes were Found and Why?**

While it may seem self-evident that smaller animals are harder to spot than larger ones, the size bias was actually very modest: snakes spotted were, on average, merely 23 mm larger than those missed. This corroborates previous results showing that visually searching for brown treesnakes is a method far less prone to size bias than trapping (Rodda et al. 2007). Had we searched a habitat where the visibility was greater, allowing animals to be spotted from farther away, the size bias might have been more pronounced. This is simply because the smaller an object, the more difficult to spot it from a long distance.

As expected, the snake detection rate declined rapidly with distance from the transect. However, vegetation density seems not to be the sole reason for this: recall that 95% of the snakes were visible from at least some point along the transect and almost all snakes within 1 m from the transect were visible from multiple transect points. Spotting snakes perched far into the vegetation is likely more difficult at night than during broad daylight (when our data on visibility was collected). This is because the illumination level of objects close to the searcher relative to that of distant objects differs substantially at night (but not during daytime), and spotting a weakly illuminated, distant animal through a layer of nearby, brightly illuminated leaves might be difficult.

Floodlight beams allowed us to detect snakes at a somewhat wider vertical range compared to searchers using spotlights, who detected a greater concentration around eye height (Figure 3). Most striking, however, is the low detection rates of

snakes on or close to the ground, regardless of lamp type. This could in part be explained by the presence of herbaceous plants along part of the transect line, obscuring snakes at ground level. However, when watching the observers search for snakes, we also noted that they tended to spend only a small proportion of time scanning the ground. We saw searchers almost stepping on snakes sitting on the ground immediately by the transect line without noticing them. It appeared that observers were biased in directing their search efforts towards higher strata. Despite their common name, we know that brown treesnakes often move on the ground (Rodda 1989). This biased search pattern may, therefore, lead us to miss snakes that would be easy to detect, should we just look for them in the appropriate stratum. Judging from the mean perch height of snakes detected by our eight observers, they seemed to have reasonably similar search patterns (Figure 4).

Searchers' ability to find snakes declined over time. This drop in detection rate was large enough to be relevant in an operational context. While we have no certain answer to why this pattern was seen, we suspect the searchers were more observant when faced with a novel set-up coupled with the perception that the trials might be seen as a test of their ability to find as many snakes as their fellow searchers. Over time, they realized that they would not be able to find more than a small fraction of the snakes (and that being true also for the other searchers); thus they may have become more relaxed and less observant to the extent that the detection rate was negatively affected. This indicates some level of positive competition among searchers or perhaps a reward for finding snakes (Henke 1998) may be needed to maintain the highest possible detection rate during extended search efforts.

### How Good Can we Afford to Get?

In the best of worlds, we would not have to trade one good lamp trait for another. Given ample power supply we might opt for a lamp that has both a wide beam and a high light intensity. The 20W floodlight used in this study match those criteria. With a 9Ah NiMH battery pack, weighing about 600 g, it lasts for about 2 hr 15 min in the 20W mode. Two battery packs (costing approximately US $125 each) are thus appropriate for an evening's field work. Those aiming for the best possible detection rates might aim for an even more powerful metal halide or LED headlamp. Marketed for orienteering and mountain biking, these come with re-chargeable lithium-ion batteries. While their wattages are typically similar to or somewhat lower than the halogen lamp we used, the amount of light emitted per watt is considerably higher – as is, unfortunately, the price tag. Even though the price of high-power floodlight headlamps might seem daunting, it is small compared to the cost of labor for extended search efforts. When used regularly, the price paid per snake detected is probably lower for high-end headlamps than cheaper models. Considering the possibility that high snake detection rate may be the difference between BTS succeeding in or failing to colonize another island, the economic benefits of using the best possible lamp increases even more.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

Bearder, S. K., K. A. Nekaris, and D. J. Curtis. 2006. A re-evaluation of the role of vision in the activity and communication of nocturnal primates. Folia Primatologica 77:50-71.

Buckland, S. T., D. R. Anderson, K. P. Burnham, and J. L. Laake. 1993. Distance sampling: estimating abundance of biological populations. Chapman and Hall, London.

Burnham, K. P., and D. R. Anderson. 2002. Model selection and multimodel inference: a practical information-theoretic approach. 2nd edition. Springer-Verlag, New York, USA.

ENGEMAN, R. M., M. A. LINNELL, D. S. VICE, AND M. E. PITZLER. 1998. Efficacy of the methods used in an integrated program to deter the dispersal of BTS from Guam. Pages 435-440 *in* G. Pickles, editor. 11th Australian Vertebrate Pest Conference, Bunbury, Western Australia, 3-8 May 1998. Programme and Proceedings.

FRITTS, T. H. 1987. Movements of snakes via cargo in the Pacific region. 'Elepaio 47:17-18.

GURKA, M. J. 2006. Selecting the best linear mixed model under REML. The American Statistician 60:19-26.

HENDERSON, R. W. 2002. Neotropical treeboas - natural history of the *Corallus hortulanus* complex. Krieger, Malabar, Florida, USA.

HENKE, S. E. 1998. The effect of multiple search items and abundance on the efficiency of human searchers. Journal of Herpetology 32:112-115.

KRAUS, F., AND D. CRAVALHO. 2001. The risk to Hawai'i from snakes. Pacific Science 55:409-417.

MCCOID, M. J., AND D. W. STINSON. 1991. Recent snake sightings in the Mariana Islands. 'Elepaio 51:6.

RIBI, W. A. 1981. The phenomenon of eye glow. Endeavour 5:2-8.

RODDA, G. H. 1989. Foraging behaviour of the brown treesnake, *Boiga irregularis*. Herpetological Journal 2:110-114.

RODDA, G. H., AND T. H. FRITTS. 1992. The impact of the introduction of the Colubrid snake *Boiga irregularis* on Guam's lizards. Journal of Herpetology 26:166-174.

RODDA, G. H., J. A. SAVIDGE, C. L. TYRRELL, M. T. CHRISTY, AND A. R. ELLINGSON. 2007. Size bias in visual searcher and trapping of brown treesnakes on Guam. Journal of Wildlife Management 71:656-661.

SAS INSTITUTE. 2003. SAS/STAT software version 9.1, SAS Institute, Inc., Cary, North Carolina, USA.

SAVIDGE, J. A. 1987a. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660-668.

SAVIDGE, J. A. 1987b. The ecological and economic impacts of an introduced snake on Guam and its threat to other Pacific islands. Ples 3:29-34.

SOMIYA, H. 1980. Fishes with eye shine – functional morphology of guanine type *tapetum lucidum*. Marine Ecology Progress Series 2:9-26.

STAVENGA, D. G., J. A. J. NUMAN, J. TINBERGEN, AND J. W. KUIPER. 1977. Insect pupil mechanisms. Pigment migration in retinula cells of butterflies. Journal of Comparative Physiology 113:73-93.

# CAN TEMPERATURE BE USED AS A TOOL FOR LIMITING BROWN TREESNAKE INVASION VIA TRANSPORTATION PATHWAYS?

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***Abstract:*** The use of extreme temperatures is one of the few chemical residue-free techniques available for control of pest species and has proven fast and effective in a variety of applications. We determined the upper and lower lethal temperatures for the brown treesnake. We also investigated whether gender, size, body condition and exposure time influenced survival at temperature extremes. Elevating snake core temperatures to 40°C and 41°C for one hour resulted in 99% mortality within seven days (40°C) and 100% mortality within one hour of exposure (41°C). Mortality decreased sharply to 51% for a one hour exposure to 39°C. Shorter, heavier snakes were more susceptible to extreme hot temperatures, but gender had no effect. The lower lethal temperature was established at a body temperature of -5°C held for one hour. Males were more susceptible to extreme cold than females, as were snakes with lower body condition. These results imply that brown treesnakes are slightly more vulnerable than many invertebrate pests, suggesting that hot thermal fumigation would not require exceptional treatments and should be feasible for shipments containing most non-biological materials and some biological ones (e.g., some live plants, grain), and cold thermal fumigation should be feasible for all materials tolerant of being frozen.

***Key Words:*** *Boiga irregularis*, brown treesnake, control, Guam, invasive species, lethal, thermal tolerance, vertebrate pest.

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## INTRODUCTION

In the five decades since its accidental introduction, the brown treesnake *(Boiga irregularis*) has had an enormous negative impact on Guam's ecology and a modest detrimental effect on its economy (Savidge 1987, Fritts and McCoid 1991, Rodda and Fritts 1992, Fritts 2002). The risk of the snake spreading to other Pacific islands (Fritts et al. 1999) and there reprising its ecological devastation (Fritts and Rodda 1998) has led to an elaborate interdiction effort targeting cargo leaving Guam (Engeman et al. 1998, Vice et al. 1999). Although the current tools used in the interdiction program (trapping, visual and dog searches) enjoy reliable, albeit limited success (Engeman et al. 2002, Rodda et al. 2007), most have high associated labor costs (Vice et al. 1999, Engeman and Vice 2001, Engeman et al. 2002) and are disrupted by the frequent typhoons that strike the region (Vice and Engeman 2000, Engeman et al. 2002). Interdiction efforts would be greatly improved if effective low cost, less labor intense management tools could be developed to augment the current methods.

A variety of new snake control tools have been explored in recent years, such as ingested, dermal and aerosol toxicants (Brooks et al.1998a, Brooks et al.1998, Brooks et al.1998c, Savarie and Bruggers 1999, Savarie et al. 2001). Chemical fumigation for cargo is effective for brown treesnakes, and several products have been registered with the United States (US) Environmental Protection Agency for this purpose (Savarie et al. 2005). However, chemical fumigation is not without problems; for example, methyl bromide is hazardous for the Earth's ozone layer (Brooks et al. 1998a, Brooks et al. 1998b, Clark and Shivik 2002). Moreover, some fumigants, such as calcium cyanide and magnesium phosphide, pose a risk to human and environmental health and safety (Savarie and Bruggers 1999, Fields and White 2001, Clark and Shivik 2002), and chemicals may react adversely with certain cargos.

The use of lowered or elevated temperatures as a non-toxic, non-reactive alternative to chemical fumigation has long been recognized as an effective strategy for invertebrate pest management (e.g., Dean 1911, Dermott and Evans 1978, Fields 1992, Dowdy and Fields 2002, Mourier and Poulsen 2000, Wright et al. 2002). However, manipulating temperature for control of vertebrates is virtually untested. Given information about thermal tolerance of *B. irregularis* and temperatures in potentially snake-infested spaces such as cargo holds or aircraft wheel wells, we can predict snake survival in that space and possibly eradicate stowaways by reducing or increasing temperature beyond their thermal tolerance.

In the only study to date that addresses *B. irregularis* thermal lethal tolerance limits, Anderson and Hetherington (2000) found that anesthetized *B. irregularis* < 170 g died when exposed to 40°C for one hour. Typically, wild-caught *B. irregularis* tolerate substantial variation in environmental temperature ($T_e$) (Perry 1999, Anderson 2002, Anderson et al. 2005), but have little ability to regulate body temperature ($T_b$) when heat stressed, except to seek shade (Johnson 1975). Anderson (2002) reported mean $T_b$ of 26°C for free-ranging *B. irregularis* on Guam and suggested that the species prefers to maintain $T_b$ below 35 - 36°C in the wild. *Boiga irregularis* in the laboratory has survived temperatures up to 40°C (G. Perry, Ohio State University, unpublished data). Few studies have addressed lower lethal temperatures for reptiles other than those that hibernate through sub-freezing winters (McConnachie et al. 2007). Therefore, the minimum temperature tolerance of a tropical species like *B. irregularis* is unknown.

The role of gender, size or body condition in tolerance or recovery from extreme thermal stress is also untested in *B. irregularis*. Small sample size limited the ability of Anderson and Hetherington (2000) to determine whether mass, gender, or other related factors affected thermal resistance at high temperature. Voluntary thermal maxima of *B. irregularis* in a gradient ranged from 34.1°C to 38.5°C over a 90-min period for two snakes (860 mm SVL and 1406 mm SVL); the larger snake maintained a higher voluntary temperature than the smaller (Johnson 1975). Studies on other snakes have found conflicting results in terms of body size and thermal tolerance (e.g., Spellerberg 1972, Sanders and Jacob 1981, Doughty 1994).

Exposure time to high or low temperatures also influences the likelihood of survival (Kour and Hutchison 1970, Mahroof et al. 2003). For example, insect pests died if exposed to 40°C for more than 24 h, 45°C for 12 h, 50°C for 5 min or 55°C for 1 min (Fields 1992). Even if cargo containers and aircraft wheel wells reach temperatures that are potentially lethal to *B. irregularis*, they may not maintain such temperatures for an adequate time period. Several studies have speculated that most species of snake will not survive exposure to temperatures between 40 and 43°C for more than 0.5 h (Cowles and Bogert 1944, Brattstrom 1965, Johnson 1975), and the ability of most vertebrates to tolerate $T_b$ above 45°C is limited due to the biochemical constraint of protein denaturizing (Bartholomew 1982). Anderson and Hetherington (2000) suggested that at temperatures above 41° C, *B. irregularis* died after exposure times between 17 and 103 min, although results were variable.

Here we determine the upper ($T_{UL}$) and lower ($T_{LL}$) lethal temperatures for *B. irregularis* and discuss their utility for controlling stow-away brown treesnakes in shipping containers, aircraft cargo holds and aircraft wheel wells. We also investigate whether gender, body size and body condition can be used to predict the differential mortality of brown treesnakes exposed to high and low temperature extremes.

## METHODS

We conducted thermal trials at the US Geological Survey Laboratory at Ritidian Point, Guam, between August 2005 and September 2006. Snakes were captured in traps throughout Guam and housed individually in plastic cages (400 × 250 × 100 mm) prior to and following thermal trials. We placed cages in a covered outdoor area to provide typical ambient Guam weather conditions prior to testing, and to avoid acclimatization to the air-conditioned lab (Hutchison and Rowlan 1975, Ludderschmidt and Hutchison 1997). We furnished cages with water and refugia, checked snakes daily and fed them weekly prior to thermal trials. We maintained individuals in captivity for no longer than three weeks pre-trial. On arrival at the lab, we weighed, measured (snout-vent length [SVL] and total length) and checked each snake for general health (such as cuts, abrasions, cloudy eyes, lethargy and general body condition). Any *B. irregularis* in ecdysis, possessing palpable follicles or considered unhealthy was rejected for use in trials. We determined gender by probing or everting hemipenes.

**Thermal Chamber**

We used an Envirotronics™ (Model SH8) environmental chamber to test B. irregularis thermal tolerance. The chamber (internal size: 0.11 $m^3$) had an accuracy of ± 0.5°C at -10°C to 50°C, and ± 2% relative humidity (RH) at 40 - 90% RH, above and below which accuracy decreased (± 1.0°C or ± 5% RH). During trials, each snake was individually housed in galvanized metal mesh and clear plastic snake compartments (300 × 150 × 150 mm for snakes < 1,500 mm SVL and 445 × 385 × 175 mm for snakes > 1,500 mm SVL) in the chamber. We tested four (if <1,500 mm SVL) or two (if > 1,500 mm SVL) snakes per trial. For $T_{UL}$ trials, absolute humidity in the chamber was held constant from a starting point of 25°C and 75% RH. RH was set to 0% at the target temperature for $T_{LL}$ trials, although this value is not realistically achievable for very low temperatures by the chamber. Hence, we recorded actual RH attained during trials.

**Critical Thermal Maximum and Minimum or Lethal Temperature**

Critical thermal maxima ($CT_{Max}$) or minima ($CT_{Min}$) are commonly used in thermal studies and determined by heating or cooling an animal at a standard rate that does not allow for thermal acclimation (Lutterschmidt and Hutchison 1997). The end point is usually defined as a temperature at which locomotory activity becomes disorganized and the animal loses its ability to escape from conditions that will imminently lead to its death (Cowles and Bogert 1944). Disorganized locomotion is typically defined by the loss of righting response or the sudden onset of muscular spasms (Lutterschmidt and Hutchison 1997). The lethal temperature for an animal is generally obtained by placing an animal in a constant temperature environment and duration of exposure until death is recorded (Hutchison and Rowlan 1975, Huey 1982, Eckert and Randall 1983). Our experiment draws from both methods in that we heated or cooled snakes at a standard rate up to the focal temperature and ran trials over a set time period after core body temperature equilibration. Our dependent variable was survival, but the duration of exposure was fixed, not contingent on snake behavior. Our research goal was to identify a management prescription that did not require fumigators to observe snake behavior, as the intended application was to kill *unseen* snakes (detected snakes would have already been removed from the cargo). It was not possible to predict $T_{UL}/T_{LL}$ from published $CT_{Max}/CT_{Min}$ values or vice versa, as a variety of methods and end points have been applied in different studies, those end points have not always been defined consistently or unequivocally, and because of the variability in other factors such as size and gender that could affect thermal robustness after CT is reached (Hutchison and Rowlan 1975, Lutterschmidt and Hutchison 1997, McConnachie et al. 2007).

Due to unknown variability in the thermal inertia of materials that would be operationally exposed to thermal fumigation, we chose to concentrate on snake tolerance to critical temperature for a fixed exposure time after a snake had equilibrated to the treatment temperature. Operational application will thus require use of thermal sensors to monitor the heating or cooling rate of the treated materials and "start the treatment clock" only after equilibration is complete. The proposed management prescription is based on one hour of exposure after the treated material has reached (equilibrated to) treatment temperature. Because cargos are vastly larger (and therefore have more thermal inertia) than snakes, the time required for cargo equilibration is expected to be uniformly sufficient to ensure that any enclosed snakes have also equilibrated to the target temperature before the fixed treatment period begins.

Our experimental heat treatment sequence involved four periods: (1) ramp up, (2) equilibration, (3) heat treatment, and (4) ramp down. Experimental cold treatments were the converse: (1) ramp down, (2) equilibration, (3) cold treatment, and (4) ramp up. The starting and ending points were always 25° C at 75% RH. Ramp-up or ramp-down rates were fixed at a standard rate of 1° C $min^{-1}$ (Lutterschmidt and Hutchison 1997). At the conclusion of each trial the animals were left in the chamber at 25° C for five minutes prior to removal. The animals treated in any given trial were of approximately the same size, so that they would equilibrate at about the same rate; the duration of the equilibration period was set to ensure that each snake would reach the target temperature shortly before the beginning of the treatment period.

**Estimating Requisite Equilibration Duration**

Heat transfer is dependent on body size and shape (Lutterschmidt and Hutchison 1997, Seebacher and Shine 2004). Copper tube models were initially used to estimate the time required for $T_b$ and $T_e$ to equilibrate. We constructed models from hollow copper pipes of three different diameters (12.7, 19.05 and 25.4 mm) that most closely represented the circumference of snakes tested. We cut pipes into 500 mm lengths which were filled with water and sealed with copper caps. When filled with water, the small, medium and large models weighed 64, 145 and 257 g respectively. A small hole was drilled into the side of each model through which the thermocouple probe from a pre-calibrated temperature Hobo™ data logger (Onset Computer Corporation, accuracy ± 1.0°C from -30°C to 50°C) was threaded. Data-loggers recorded temperature every 60 sec throughout each trial.

To determine the actual duration of $T_b/T_e$ equilibration and $T_b$ throughout the trials, we fed temperature loggers (i-button™; Maxim Integrated Products, 16 × 5 mm; ± 1.0°C from 30°C - 70°C) to at least 25% of *B. irregularis* tested at each temperature. We also attached an additional i-button inside each snake compartment to record $T_e$ and placed one Hobo™ temperature and humidity logger (accuracy ± 3% RH from 0 - 50°C) in the chamber.

**Trial Set Up**

We weighed to the nearest gram (g), measured SVL (mm) and recorded heart rate (bpm) of each snake immediately prior to placement in the chamber. Snakes were used in one trial only. At the completion of each trial we recorded heart rate, breathing rate (breaths per five min period), mass and pupil constriction (1 = constricted, 2 = normal, 3 = dilated). If snakes were alive, we recorded whether they were able to right themselves within 5 min. Live snakes were returned to their cages and observed daily for seven days. We euthanized all snakes that survived seven days post trial and confirmed gender and presence/absence of food in the gut via dissection. We determined $T_{UL}$ and $T_{LL}$ had been reached when a snake's $T_b$ attained the test temperature for one hour and the snake died within seven days post trial.

**Sequence of Test Temperatures**

We initially tested $T_{UL}$ at 39°C, based on existing literature (Anderson and Hetherington 2000) and unpublished data. Following the 39°C trial, our protocol was to home in on the lethal threshold by increasing or decreasing temperature by 1° C increments depending on the outcome of the initial trial. Our protocol for determining $T_{LL}$ was similar, although setting our initial $T_{LL}$ was more difficult because of lack of available data. However, based on results for other reptiles (Claussen et al. 1990, McConnachie et al. 2007), anecdotal information and informed estimates, we initiated our $T_{LL}$ test at -3°C.

**Individual Snake Attributes**

Our initial trials aimed to test a group of snakes we considered most likely to survive exposure to extreme temperatures. Based on literature including but not limited to Johnson (1975), Huey (1982), Gibson et al. (1989), Lutterschmidt and Reinert (1990), Peterson et al. (1993), Madsen and Shine (1996), Anderson and Hetherington (2000), Anderson (2002), and Ladyman and Bradshaw (2003), we hypothesized that large (>1,000 mm SVL), sexually mature, non-shedding, non-follicular (if female), fully hydrated and satiated *B. irregularis* with a high (>1.0) body condition index were the most likely to survive extreme temperatures. Because snakes better tolerate cold air when it is dry (McConnachie et al. 2007), we did not provide water during trials. We countered the potential drying effect of the hot temperature trials with elevated humidity (see above). Snakes entered the chamber having consumed food item(s) of a size approximately 10% of their body weight between 48-72 h prior to the trial (e.g., Secor and Diamond 1997).

Following the initial trials to identify critical temperature ranges, we tested $T_{UL}$ and $T_{LL}$ for *B. irregularis* of different size, gender and body condition. We tested the effect of gender of snakes assigned to one of five size classes for each trial (≥50, 50–100, 100–150, 150–200, and 200–250 g). To balance potential gender effects, we maintained roughly an equal sex ratio for each size class. Unless a snake was a neonate or juvenile (i.e., <600 mm SVL), individuals of known sex were tested. We also tested an additional group of *B. irregularis* that were exposed to $T_{UL}$ for 0.5 h (instead of 1 h) following equilibration.

**Analysis**

We evaluated the relationship of gender and three body metrics (condition index, SVL, and mass) to survival. Condition index was calculated as the ratio of a snake's individual mass to its expected mass given its length. Expected mass was

estimated by linear regression on a logarithmic scale, and based on the 286 snakes used in this study. We calculated the means, standard deviation, range, and 95% confidence limits (CL) for snakes of each thermal treatment and used 95% CL to assess differences between groups. Unless otherwise stated, values expressed are mean ± standard deviation. The Chi-square statistic was used to compare temperature tolerance between genders. Statistical significance was set at $\alpha \leq 0.05$.

## RESULTS

### Validation of Chamber Temperature

We placed thermocouples (hobo loggers and i-buttons) within the chamber to verify accuracy of the device. At set temperatures above 38°C during 97 trials, the actual chamber temperature was on average 0.67°C below the set temperature ($\bar{x}_{set}$ = 38.99 ± 0.05°C, $\bar{x}_{actual}$ = 38.32 ± 0.19°C; Figure 1). To account for the difference, we set the chamber temperature 0.5°C higher for all trials.

### Determining $T_b/T_e$ Relationship

The larger the copper model, the greater the time to $T_b/T_e$ equilibration ($r^2$= 0.89; Figure 2). Models heated to 41°C took approximately 10 min longer to reach equilibrium than those heated to 38°C regardless of size (64 g = 13–22 min; 145 g = 22–32 min; 257 g = 33–42 min). We used the regression equation derived from the copper model trials to estimate the equilibration time required for *B. irregularis* of varying sizes at 40 and 41°C and compared the estimated values with actual values from snakes with ingested i-buttons. We found that the copper models consistently overestimated the time required for a snake $T_b$ to equilibrate by an average of 10 min (± 6.48 min, $n$ = 22). However, our primary goal was to ensure that a snake's $T_b$ was held at the test temperature for a minimum of one hour, thus slightly overestimating the time did not compromise the experimental design.

### $T_{UL}$

We tested 233 snakes over three hot temperatures (39°C: $n$ = 92, 40°C: $n$ = 88 and 41°C: $n$ = 53). Forty nine percent of snakes exposed to 39°C survived (Figure 3). Survival decreased to 99% at 40°C (within 7 days) and 0% at 41°C, the temperature at which no snake survived longer than 1 h post treatment.

At 39°C, there was no difference in susceptibility to death between genders ($\chi^2$ = 0.70, DF = 1, $P$ = 0.4). Of the three body metrics assessed, SVL and mass were better indicators of susceptibility to death than condition index, although confidence intervals for both of those metrics overlapped (Table 1). On average, shorter, heavier snakes were more likely to die after 1 h exposure to 39°C than longer, lighter snakes.

When exposure time was halved from 1 h to 0.5 h, the death rate at 1 h and 1 d post-treatment decreased marginally (Figure 3). However, overall mortality was similar between the two treatments (99% for 1 h exposure, 95% for 0.5 h exposure). Snakes with better body condition exposed to 40°C for 0.5 h died just as readily as those exposed to 40° for 1 h ($\bar{x}_{0.5\,h}$ = 1.12 ± 0.16; $\bar{x}_{1\,h}$ = 0.99 ± 0.15). Neither length ($\bar{x}_{0.5\,h}$ = 996 ± 218 mm SVL; $\bar{x}_{1\,h}$ = 1000 ± 273 mm SVL) nor mass ($\bar{x}_{0.5\,h}$ = 144.6 ± 75.3 g; $\bar{x}_{1\,h}$ = 160.9 ± 168.0 g) affected survival at either exposure period.

### $T_{LL}$

We tested 53 snakes at two cold temperatures: -3°C ($n$ = 17) and -5°C ($n$ = 36). A larger sample size was initially planned; however, because of ethical considerations resulting from the unanticipated protracted time to death, we chose to reduce the number of low-temperature trials. All snakes died within 7 d of exposure to -5°C, but 29% survived past 7 d at -3°C (Figure 4). At both temperatures most deaths occurred between 24 h and 1 d post-trial. At -3°C, male snakes were more susceptible to death than females ($\chi^2$ = 4.41, DF = 1, $P$ = 0.04). Snakes that died at -3°C had less body mass than those that survived, although confidence intervals overlapped (Table 1).

## DISCUSSION

The aim of our study was to quantify the high and low lethal temperatures of the snake *B. irregularis* to evaluate possible use of thermal manipulation as an interdiction tool for snakes in cargo. The upper and lower thermal limits resulting in mortality for one hour exposure after equilibration (-5 and 41 C) were values similar to those that are already in use for thermal fumigation of insect pests (Fields 1992, Mourier and Poulsen 2000, Wright et al. 2002). Thus, thermal fumigation can be used for both

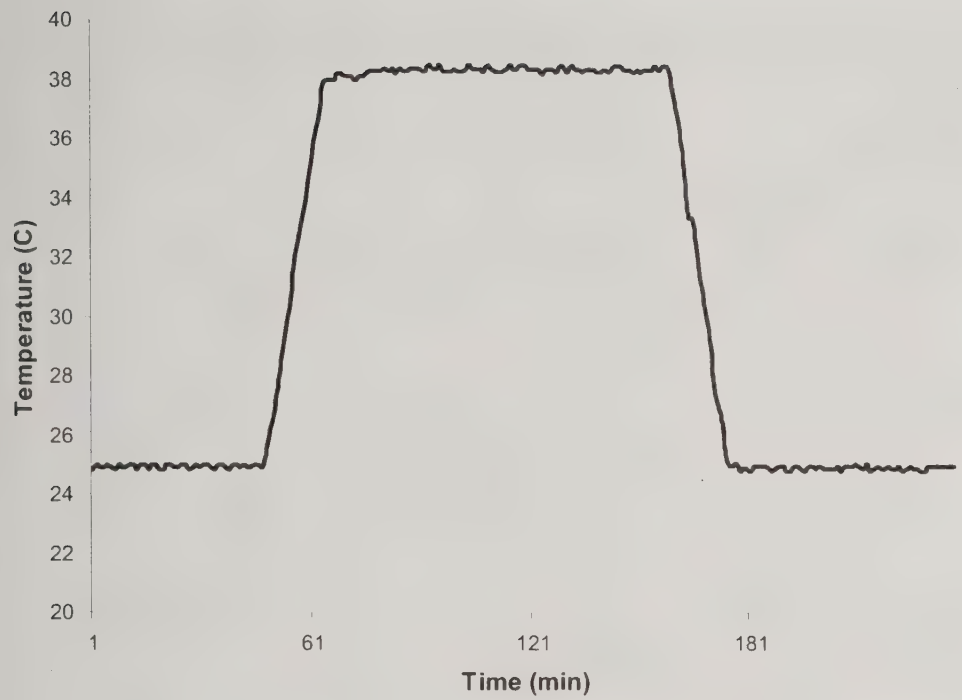



**Figure 1.** Comparison between the set chamber temperature (grey line) and the actual chamber temperature (black line) for a 39°C trial. The chamber was set to 25°C after which temperature increased to 39°C at 1°C · $min^{-1}$ before returning to 25°C at the same cooling rate. The set temperature was subsequently adjusted upward by 0.5°C to compensate for the difference between set and actual chamber temperature.

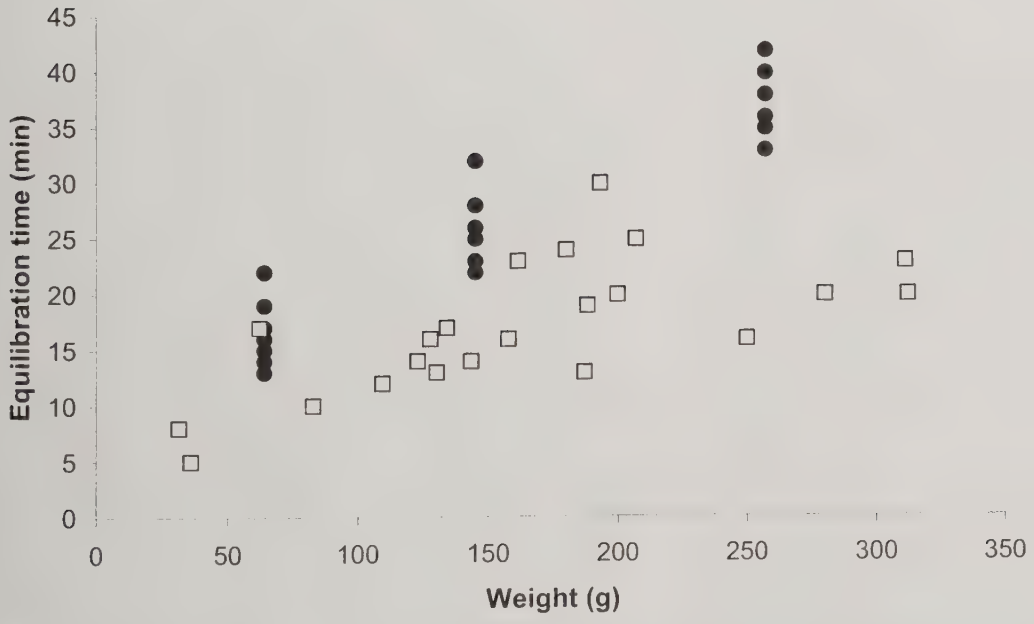



**Figure 2.** Copper models (●) consistently overestimated the time *Boiga irregularis* (□) required to equilibrate to $T_c$, as indicated by internal temperature loggers. A group of six copper models of three sizes (64, 145 and 257 g) were run twice at four temperatures (38 - 41°C). *B. irregularis* were tested at 40 - 41°C ($n$ = 22 snakes).

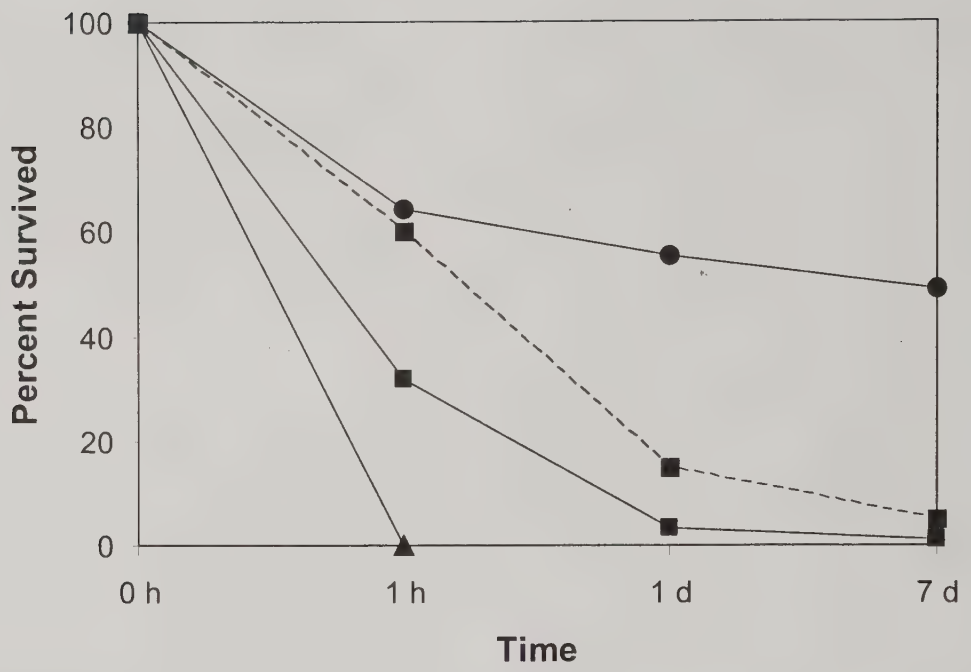



**Figure 3.** The number of *Boiga irregularis* that died within 1 h, 1 d or 7 d after exposure to temperatures of 39°C (●), 40°C (■) and 41°C (▲) for 1 hour plus equilibration time. The dashed line represents 0.5 h exposure to 40°C (--■--). At 1 h exposure, all snakes died within an hour at 41°C ($n$ = 53 snakes), only one of 88 snakes tested at 40°C ultimately survived, but 49% survived past 7 days when the temperature was lowered to 39°C ($n$ = 92 snakes). When exposure time was decreased to 0.5h, 5% survived at 40°C ($n$ = 20 snakes).

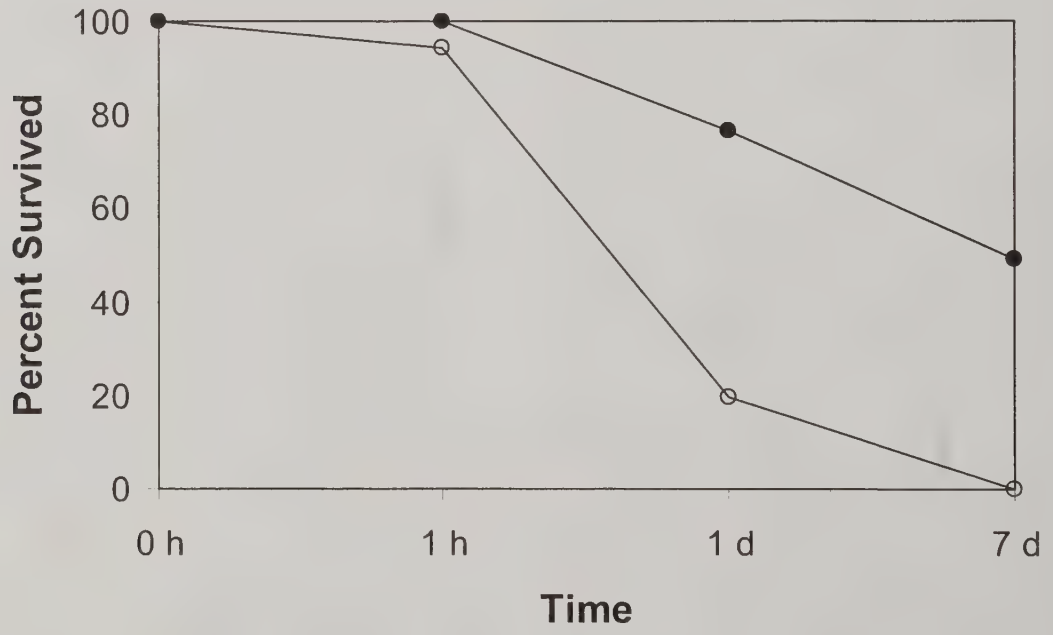



**Figure 4.** The number of *Boiga irregularis* that died within 1 h, 24 h or 7 days after exposure to -3°C (●) and -5°C (○) for 1 hour plus equilibration time. All snakes died within seven days of exposure to -5°C ($n$ = 36 snakes) but 49% survived past 7 days when exposed to -3°C ($n$ = 17 snakes).

**Table 1.** Mean, range and 95% confidence limits (CL) for condition index, SVL (mm) and mass (g) for 109 *Boiga irregularis* that either died or survived when exposed to 39°C or -3°C for one hour plus equilibration time. *n* = number of snakes.

| | 39°C | | -3°C | |
|---|---|---|---|---|
| | **Survived** | **Died** | **Survived** | **Died** |
| *n* | 45 | 47 | 5 | 12 |
| Condition Index | | | | |
| Mean ± SD | 1.00 ± 0.15 | 0.99 ± 0.17 | 1.10 ± 0.12 | 1.15 ± 0.14 |
| Range | 0.73 – 1.36 | 0.65 – 2.00 | 0.98 – 1.30 | 0.98 – 1.38 |
| 95% CL | 0.96 – 1.04 | 0.95 – 1.04 | 0.99 – 1.20 | 1.07 – 1.23 |
| Snout-Vent Length (mm) | | | | |
| Mean ± SD | 998.56 ± 184.80 | 947.91 ± 88.13 | 1029.00 ± 159.27 | 952.42 ± 24.68 |
| Range | 608.00 – 1205.00 | 510.00 – 1290.00 | 905.00 – 1305.00 | 852.00 – 1090.00 |
| 95% CL | 832.56 – 940.55 | 922.72 – 973.11 | 889.39 – 1168.61 | 904.04 – 1000.80 |
| Mass (g) | | | | |
| Mean ± SD | 91.61 ± 68.45 | 123.10 ± 88.13 | 151.12 ± 85.76 | 115.58 ± 41.18 |
| Range | 20.90 – 269.20 | 11.20 – 290.00 | 84.60 – 298.00 | 65.60 – 190.80 |
| 95% CL | 71.61 – 111.61 | 97.90 – 148.29 | 75.94 – 226.30 | 92.29 – 138.88 |

vertebrate and invertebrate pests with only minor modification of the treatment protocol. Many cargoes can tolerate exposure to these temperatures, but especially sensitive live materials (e.g., some horticulture materials) probably can not. Those might be candidates for chemical fumigation.

Individual variation in temperature tolerance was minor, but was not fully explained by the individual covariates we measured (gender, size or body condition). However, shorter, heavier snakes were more likely to die at one hour exposure to 39°C than longer, lighter snakes. At cold temperatures death was protracted and snakes took up to 6 days to die at -5° C; thus brown treesnakes arriving in the wheel wells of aircraft (i.e., exposed to extreme cold en route) may be active but moribund on arrival, and may cease movement many days later. The most robust group of snakes at cold temperatures was somewhat heavy-bodied females.

While we expected *B. irregularis* $T_{UL}$ to be between 40°C and 43°C, we were surprised to find that this semi-tropical snake, originating from areas where overnight winter temperatures rarely drop below 0°C, can survive sub-freezing conditions. The mechanisms that allow *B. irregularis* to survive extreme cold depend on its ability to control or prevent ice formation (Ramløv 2000), either by adaptive freeze tolerance (Storey and Storey 1996) or freeze avoidance (Costanzo et al. 1995). Freeze tolerance in reptiles is rare, and is usually limited to species that experience regular freezing of up to 65% of body fluids (e.g., Storey 1990, Costanzo et al. 1995). The ice content in *B. irregularis* was not tested, but we did observe tissue damage in snakes exposed to -5°C that we attributed to freezing. It therefore seems unlikely that the species is freeze tolerant. It is more plausible that *B. irregularis* avoids freezing by being able to supercool (the ability to depress freezing point by the production of antifreeze compounds or cryoprotectants; McConnachie et al. 2007).

Perry and Vice (1999) documented thermal conditions within loaded and unloaded cargo shipping containers stacked on a dock, and in

loaded containers in transit from Guam to Micronesia and the US mainland. Assuming a $T_{UL}$ of 41°C for one hour, containers consistently failed to reach adequate temperatures to ensure snake mortality while in transit. Lethal temperatures were reached dockside, but the capacity to attain sufficient temperatures was dependent upon shading, stacking and whether the container was full or empty. It is therefore unlikely that $T_{UL}$ for *B. irregularis* could be achieved by passive heating of containers alone. However, passively increasing temperatures in shipping containers could be a cost-effective technique to augment the efficacy of forced hot-air fumigation. Moreover, by identifying low-risk cargo containers (i.e., those that have reached 41°C throughout the entire container for at least 1 hour), only the high-risk containers (those that have not reached 41°C for at least one hour) would require searching.

In contrast to hot temperature fumigation, the use of active cold temperature fumigation has limited applications based on practicality. For example, unless the shipment is of frozen food (very low risk), the cost of increasing the temperature of a container by 11°C from 25°C to 41°C is presumably less than cooling the container by 30°C from 25°C to - 5°C, particularly if the container is passively warmed by solar radiation prior to fumigation.

$T_{LL}$ may nonetheless provide useful guidelines for management. Perry (2002) found that during flight, cargo hold temperatures averaged 18.6 °C (± 5.0 °C SD), not sufficient to cause death in *B. irregularis*. On the other hand, wheel wells of aircraft flying at high altitude frequently drop well below freezing (average minimum of -18°C) on long-haul flights (>7 h), increasing the probability of death of a wheel-well stowaway (Perry 1999, 2002). In combination with knowledge of $T_{LL}$, this information can be used to reduce inspection effort (cost) for low risk flights.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

ANDERSON, N. L. 2002. Thermal preferences, metabolic rate, and water flux of the brown treesnake (*Boiga irregularis*) in the laboratory and on Guam. Dissertation, Ohio State University, Columbus, USA.

ANDERSON, N. L., AND T. E. HETHERINGTON. 2000. Report for pilot study on determination of the minimal lethal thermal environment for brown treesnakes (*Boiga irregularis*) Report, Ohio State University, Columbus, USA.

ANDERSON, N. L., T. E. HETHERINGTON, B. COUPE, G. PERRY, J. B. WILLIAMS, AND J. LEHMAN. 2005. Thermoregulation in a nocturnal, tropical, arboreal snake. Journal of Herpetology 39:82-90.

BARTHOLOMEW G. A. 1982. Body temperature and energy metabolism. *In* M. S. Gordon, G. A. Bartholomew, A. D. Grinnell, C. B. Jorgensen, and F. N. White, editors. Animal physiology: principles and adaptations. 4th ed. MacMillan, New York, USA.

BRATTSTROM, B. H. 1965. Body temperatures of reptiles. American Midland Naturalist. 73:376-422.

BROOKS, J. E., P. J. SAVARIE, AND R. L. BRUGGERS. 1998a. The toxicity of commercial insecticide aerosol formulations to brown treesnakes. Snake 28:23-27.

BROOKS, J. E., P. J. SAVARIE, AND J. J. JOHNSTON. 1998b. The oral and dermal toxicity of selected chemicals to brown treesnakes (*Boiga irregularis*). Wildlife Research 25:427-435.

BROOKS, J. E., P. J., SAVARIE, J. J. JOHNSTON, AND R. L. BRUGGERS. 1998c. Toxicity of Pyrethrin/Ryrethroid fogger products to brown treesnakes, *Boiga irregularis*, in cargo containers. Snake 28:33-36.

CLARK, L., AND J. SHIVIK. 2002. Aerosolized essential oils and individual natural product compounds as brown treesnake repellents. Pest Management Science 58:775-783.

CLAUSSEN, D. L., M. D. TOWNSLEY, AND R. G. BAUSCH. 1990. Supercooling and freeze-tolerance in the European wall lizard, *Podarcis muralis*, with a revisional history of the discovery of freeze-tolerance in vertebrates. Journal Comparative Physiology B 160:137-143.

COSTANZO, J. P., C. GRENOT, AND R. E. LEE. 1995 Supercooling, ice inoculation and freeze tolerance in the European common lizard, *Lacerta vivipara*. Journal Comparative Physiology B 165:238-244.

COWLES, R. B., AND C. M. BOGERT. 1944. A preliminary study of the thermal requirements of desert reptiles. Bulletin of American Museum of Natural History 83:261-296.

DEAN, D. A. 1911. Heat as a means of controlling mill insects. Journal of Economical Entomology 4:142-58.

DERMOTT, T. AND D. E. EVANS. 1978. An evaluation of fluidized-bed heating as a means of disinfesting wheat. Journal of Stored Products Research 14:1-12.

DOUGHTY, P. 1994. Critical thermal minima of garter snakes (*Thamnophis*) depend on species and body size. Copeia 1994:537-540.

DOWDY, A. K. AND P. G. FIELDS. 2002. Heat combined with diatomaceous earth to control the confused flour beetle (Coleoptera: Tenebrionidae) in a flour mill. Journal of Stored Products Research 38:11-22.

ECKERT, R., AND D. RANDALL. 1983. Animal physiology: mechanisms and adaptations. W.H. Freeman and Company, San Francisco, California, USA.

ENGEMAN, R. M., M. A. LINNELL, D. S. VICE, AND M. E. PITZLER. 1998. Efficacy of the methods used in an integrated program to deter the dispersal of brown treesnakes from Guam. Proceeding of Australian Vertebrate Pest Conference 11:435-440.

ENGEMAN, R. M. AND D. S. VICE. 2001. Objectives and integrated approaches for the control of brown treesnakes. Integrated Pest Management Reviews 6:59-76.

ENGEMAN, R. M., VICE, D. S., D. YORK, AND K. S. GRUVER. 2002. Sustained evaluation of the effectiveness of detector dogs for locating brown treesnakes in cargo outbound from Guam. International Biodeterioration and Biodegradation. 49:101-106.

FIELDS, P. G. 1992. The control of stored-product insects and mites with extreme temperatures. Journal of Stored Products Research 28:89-118.

FIELDS, P. G., AND N. D. G. WHITE. 2001. Alternatives to methyl bromide treatments for stored-product and quarantine insects. Annual Review of Entomology 47:331-359.

FRITTS, T. H. 2002. Economic costs of electrical system instability and power outages caused by snakes on the island of Guam. International Biodeterioration and Diodegradation 49:93-100.

FRITTS, T. H., AND M. J. MCCOID. 1991. Predation by the brown treesnake *Boiga irregularis* on poultry and other domesticated animals in Guam. Snake 23:75-80.

FRITTS, T. H., M. J. MCCOID, AND D. M. GOMEZ. 1999. Dispersal of snakes to extralimital islands: incidents of the brown treesnake (*Boiga irregularis*) dispersing to islands in ships and aircraft. Pages 209-223 *in* G. H Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

FRITTS, T. H., AND G. H. RODDA. 1998. The role of introduced species in the degradation of island ecosystems: a case history of Guam. Annual Review of Ecology and Systematics 29:113–140.

GIBSON, A. R., D. A. SMUCNY, AND J. KOLLAR. 1989. The effects of feeding and ecdysis on temperature selection by young garter snakes in a simple thermal mosaic. Canadian J. Zoology 67:19-23.

HUEY, R. B. 1982. Temperature, physiology and the ecology of reptiles. Pages 25-91 *in* C. Gans, editor. Biology of the Reptilia. Academic Press, London, United Kingdom.

HUTCHISON, V. H., AND S. D. ROWLAN. 1975. Thermal acclimation and tolerance in the mudpuppy, *Necturus maculosus*. Journal of Herpetology 9:367-368.

JOHNSON, C. R. 1975. Thermoregulation in the Papuan-New Guinean boid and colubrid snakes, *Candoia carinata, Candoia aspera* and *Boiga irregularis*. Zoologica Journal of the Linnaean Society 56:283-290.

KOUR, E. L., AND V. H. HUTCHISON. 1970. Critical thermal tolerances and heating and cooling rates of lizards from diverse habitats. Copeia 1970 219-229.

LADYMAN, M., AND D. BRADSHAW. 2003. The influence of dehydration on the thermal preferences of the western tiger snake, *Notechis scutatus*. Journal of Comparative Physiology B 173:239-246.

LUTTERSCHMIDT, W. I., AND V. H. HUTCHISON. 1997. The critical thermal maximum: history and critique. Canadian Journal of Zoology 75:1561-1574.

LUTTERSCHMIDT, W. I., AND H. K. REINERT. 1990. The effect of ingested transmitters upon the temperature preference of the northern water snake *Nerodia-sipedon-sipedon*. Herpetologica 46:39-42.

MADSEN, T., AND R. SHINE. 1996. Determinants of reproductive output in female water pythons (*Liasis fuscus*: Pythonidae). Herpetologica 52:146-159.

MAHROOF, R., B. SUBRAMANYAM, J. E. THRONE, AND A. MENOND. 2003. Time-mortality relationships for *Tribolium castaneum* (Coleoptera: Tenebrionidae) life stages exposed to elevated temperatures. Journal of Economic Entomology 96:1345-1351.

MCCONNACHIE, S., G. J. ALEXANDER, AND M. J. WHITING. 2007. Lower temperature tolerance in the temperate, ambush foraging lizard *Pseudocordylus melanotus*. Journal of Thermal Biology 32:66-71.

MOURIER, H. AND K. P. POULSEN. 2000. Control of insects and mites in grain using a high temperature-short-time (HTST) technique. Journal of Stored Products Research 36:309-318.

PERRY, G. 1999. Proposal for thermal control of the brown treesnake. United States Geological Survey Biological Resources Division, Fort Collins, Colorado, USA.

PERRY, G. 2002. Wheel-well and cargo compartment temperatures of large aircraft in flight: implications for stowaways. Aviation, Space, and Environmental Medicine 73:673-676.

PERRY, G., AND D. S. VICE. 1999. An evaluation of passive thermal fumigation for brown treesnake control in surface transportation from Guam. Ohio State University, Columbus, Ohio, USA.

PETERSON, C. R., A. R. GIBSON, AND M. E. DORCAS. 1993. Snake thermal ecology: the causes and consequences of body-temperature variation. Pages 241-314 *in* R. A. Seigel, and J. T. Collins, editors. Snakes: ecology and behavior. McGraw-Hill, New York, USA.

RAMLØV, H. 2000. Aspects of natural cold tolerance in ectothermic animals. Human Reproduction 15:26-46.

RODDA, G. H., AND T. H. FRITTS. 1992. The impact of the introduction of the brown treesnake, *Boiga irregularis*, on Guam's lizards. Journal of Herpetology 26:166-174.

RODDA G. H., J. A. SAVIDGE, C. L. TYRRELL, M. T. CHRISTY, AND A. R. ELLINGSON. 2007. Size bias in visual searching and trapping of brown treesnakes on Guam. Journal of Wildlife Management 71:656-661.

SANDERS, S. S., AND J. S. JACOB. 1981. Thermal ecology of the copperhead (*Agkistrodon contortrix*). Herpetologia 37:264-270.

SAVARIE, P. J., AND R. L. BRUGGERS. 1999. Candidate repellents, oral and dermal toxicants, and fumigants for brown treesnake control. Pages 417-422 *in* G. H. Rodda, Y. Sawai, D. Chiszar and H. Tanaka, editors. Problem snake management: the habu and the brown treesnake. Cornell University Press, Ithaca, New York, USA.

SAVARIE, P. J., J. A. SHIVIK, G. C. WHITE, J. C. HURLEY, AND L. CLARK. 2001. Use of acetaminophen for large-scale control of brown treesnakes. Journal of Wildlife Management 65:35-65.

SAVARIE, P. J., W. S. WOOD, G. H. RODDA, R. L. BRUGGERS, AND R. M. ENGEMAN. 2005. Effectiveness of methyl bromide as a cargo fumigant for the brown treesnake (*Boiga irregularis*). International Biodeterioration & Biodegradation 56:40-44.

SAVIDGE, J. A. 1987. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660- 668.

SECOR, S. M., AND J. DIAMOND. 1997. Effects of meal size on postprandial responses in juvenile Burmese pythons (*Python molurus*). American Journal of Physiology 272:902-912.

SEEBACHER, F., AND R. SHINE. 2004. Evaluating thermoregulation in reptiles: the fallacy of the inappropriately applied method. Physiological and Biochemical Zoology 77:688-695.

SPELLERBERG, I. F. 1972. Temperature tolerances of southeastern Australian reptiles examined in relation to reptile thermoregulatory behavior and distribution. Oecologica 9:23-46.

STOREY, K. B. 1990. Life in a frozen state: adaptive strategies for natural freeze tolerance in amphibians and reptiles. American Journal of Physiology Regulatory, Integrative and Comparative Physiology 258:559-568.

STOREY, K. B., AND J. M. STOREY. 1996. Natural freezing survival in animals. Annual Review of Ecology and Systematics 27:365-386.

VICE, D. S., M. S. LINNELL, AND M. E. PITZLER. 1999. Summary of Guam's outbound cargo handling process: preventing the spread of the brown treesnake. United States Department of Agriculture, Barrigada, Guam, USA.

VICE, D. S., AND R. M. ENGEMAN. 2000. Brown treesnake discoveries during detector dog inspections following supertyphoon Paka. Micronesica 33:105-110.

WRIGHT, E. J., E. A. SINCLAIR, AND P. C. ANNIS. 2002. Laboratory determination of the requirements for control of *Trogoderma variabile* (Coleoptera: Dermestidae) by heat. Journal of Stored Products Research. 38: 147-155

# SIZE-BASED TRENDS AND MANAGEMENT IMPLICATIONS OF MICROHABITAT UTILIZATION BY BROWN TREESNAKES, WITH AN EMPHASIS ON JUVENILE SNAKES

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***Abstract:*** The brown treesnake (*Boiga irregularis*, or BTS), a costly invasive species, has been the subject of intensive research on Guam over the past two decades. The behavior and habitat use of hatchling and juvenile snakes, however, remain largely unknown. We used a long-term dataset of BTS captures (N = 2,415) and a dataset resulting from intensive sampling within and immediately around a 5-ha fenced population (N = 2,541) to examine habitat use of BTS. Small snakes were almost exclusively arboreal and that they appeared to prefer tangantangan (*Leucaena leucocephala*) habitats. In contrast, large snakes used arboreal and terrestrial habitats in roughly equal proportion, and were less frequently found in tangantangan. Among snakes found in trees, there were no clear size-based preferences for certain heights above ground, nor for size-based choice of perch diameters. We discuss these results as they relate to management and interdiction implications for brown treesnakes on Guam and in potential incipient populations on other islands.

***Key Words:*** *Boiga irregularis*, brown treesnake, Guam, habitat use, invasive species, interdiction, Saipan, visual searching.

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## INTRODUCTION

Among invasive species worldwide, the brown treesnake (*Boiga irregularis*, or BTS) is routinely upheld as an archetypical invasive species, largely due to its decimation of the forest avifauna of Guam after its accidental introduction to the island in the 1940s (Savidge 1987, Rodda et al. 1999b). The history of this introduction and its environmental and economic effects has recently been subject to an updated review by Rodda and Savidge (2007). Despite the wide acceptance of the BTS as an exemplar of the ecological havoc that can be wreaked by a vertebrate invader, there are still many aspects of this species' biology that are poorly understood.

Among the poorly known aspects of BTS biology is the habitat use and behavior of hatchling and juvenile snakes. On Guam, BTS hatch at about 375 mm snout-vent length (SVL), and are mature at 950-1050 mm SVL (Savidge et al. 2007). While the advent of miniature radiotransmitters has revolutionized the study of secretive, cryptic snakes by allowing multiple re-locations of the same individual over time (Shine and Bonnet 2000), juvenile BTS are too small (about 4-8 g at hatching) to allow surgical implantation of transmitters with sufficient battery life for assessment of habitat use. Furthermore, until recently there was no proven means of reliably locating small BTS in complex forested environments on Guam. Traps baited with live mice (*Mus musculus*) are the primary control tool currently in use on Guam (Rodda et al 1999a, Vice et al. 2005). However, these traps are largely ineffective for snakes <850 mm SVL (Rodda et al. 2007), and thus only capture a segment of the population. In contrast, recent research indicated that visual searching by trained personnel is effective for detecting snakes across the range of BTS body sizes on Guam, including hatchlings and juveniles (Rodda et al. 2007).

Subjective impressions of various observers on Guam over the years suggest that small snakes tend towards being arboreal, while the largest snakes were more prevalent on the ground. Similarly, small snakes are often inclined to be more common in stands of introduced tangantangan (*Leucaena leucocephala*). However, these anecdotal observations have been offered by many different individuals of variable searching ability, and observations are additionally subject to variation in habitat types, temporal changes in prey availability and snake behavior, and other contributors to

variability in snake detectability. In the face of these potential biases, it is difficult to quantify differential behavior and habitat use between small (<850 mm SVL, roughly the size at which ~50% of snakes become trappable; Rodda et al. 2007) and large (≥850 mm SVL) snakes. One possible solution is to examine data from observations of BTS over many years, allowing for a large sample size and thus more rigorous analyses. However, historical datasets may be subject to the same types of observer, habitat, and temporal bias as are the anecdotal impressions mentioned above, rendering their utility somewhat doubtful.

We analyzed data from both island-wide historical and intensive site-specific visual surveys for BTS. Our goals were to: (1) assess the concordance between these types of datasets to determine whether reliable conclusions can be made from potentially biased historical data, (2) analyze habitat use of small versus large snakes, and (3) discuss management implications of these results.

## METHODS

We used two large datasets of BTS observations on Guam. We did not use any data resulting from trap captures of BTS, as these captures indicate height at which a trap was set and other trap placement characters, rather than independent instances of habitat choice by BTS. The first dataset was a historical (1988-2003) dataset of captures and observations resulting from visual surveys on Guam (hereafter referred to as the HIST dataset). This dataset comprised 2,415 visual observations of BTS detected island-wide, including targeted searches in numerous habitat types as well as incidental observations while conducting other fieldwork (black bars in Figure 1). Of these observations, 2,110 records included body size (total length) of the snake, 2,336 records included vertical height of the snake when first observed, 1,270 records included perch diameter (diameter of the perch upon which the snake was found, for arboreal individuals only), and 2,381

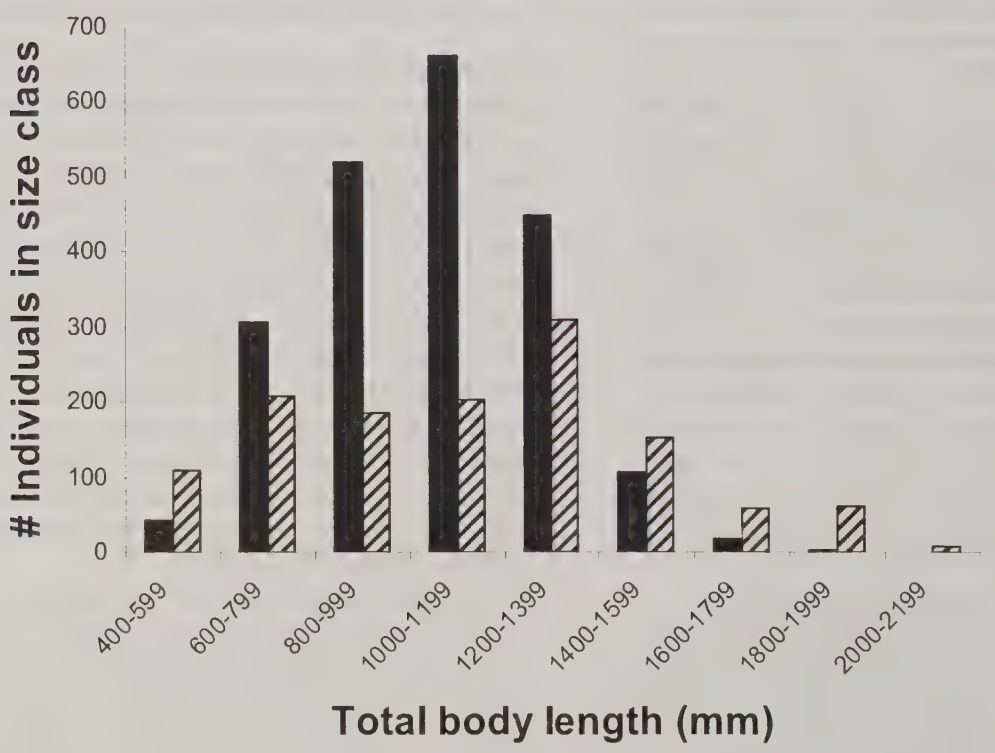



**Figure 1.** Sample sizes of BTS captures in the datasets used in analyses. Black bars indicate data from the HIST dataset, while hatched bars indicate data from the NWFN dataset.

records included the substrate on which the snake was first observed, including identification of the species of tree used by arboreal snakes. Body sizes were measured in the case of snakes that were captured, or estimated to the nearest 100 mm by experienced BTS biologists for snakes that could not be captured.

The second dataset was from intensive sampling from 2004-2007 in and immediately around a geographically closed population at Northwest Field North, on Andersen Air Force Base (Rodda et al. 2007). Briefly, the closed population is a 5-ha fenced plot that is impermeable to BTS immigration or emigration. This dataset included 2,541 observations of 328 individuals (hereafter referred to as the NWFN dataset; hatched bars in Figure 1). For the purposes of these analyses, we considered each capture of an individual to be an independent observation of habitat use, and thus we included multiple observations of individuals. Of these observations, 1,294 records included body size of the snake, 1,909 records included vertical height of the snake when first observed, 1,602 records included perch diameter (for arboreal individuals only), and 1,897 records included the type of perch, including identification of the species of tree used by arboreal snakes.

Means are reported ± one standard deviation. Data were $\log_{10}$-transformed when necessary to meet assumptions of normality in linear analyses. We used linear regression to analyze the amount of variation in perch diameter and vertical heights used by snakes as a function of snake body size.

## RESULTS

Mean total body length of BTS in the HIST dataset was 997 ± 238 mm and ranged from 400 to 1,900 mm. Over 90% of small snakes were found in trees (black bars in Figure 2), while mean length of BTS found on the ground was 1,181 ± 179 mm total length, which is ~18% larger than the overall average. Conversely, the proportion of individuals in tangantangan decreased with increasing body length (black bars in Figure 3); mean total length of BTS found in tangantangan was 900 ± 214 mm, which is ~10% smaller than the overall average. Whilst use of terrestrial habitats was dominated by large snakes and use of tangantangan by small snakes, medium-sized snakes were somewhat more likely to be found on fences (Figure 4); mean body length of snakes on fences was 1,005 ± 211 mm. Among snakes in vegetation, mean perch diameter was 18.2 ± 30.2 mm and ranged from 1 to 950 mm.

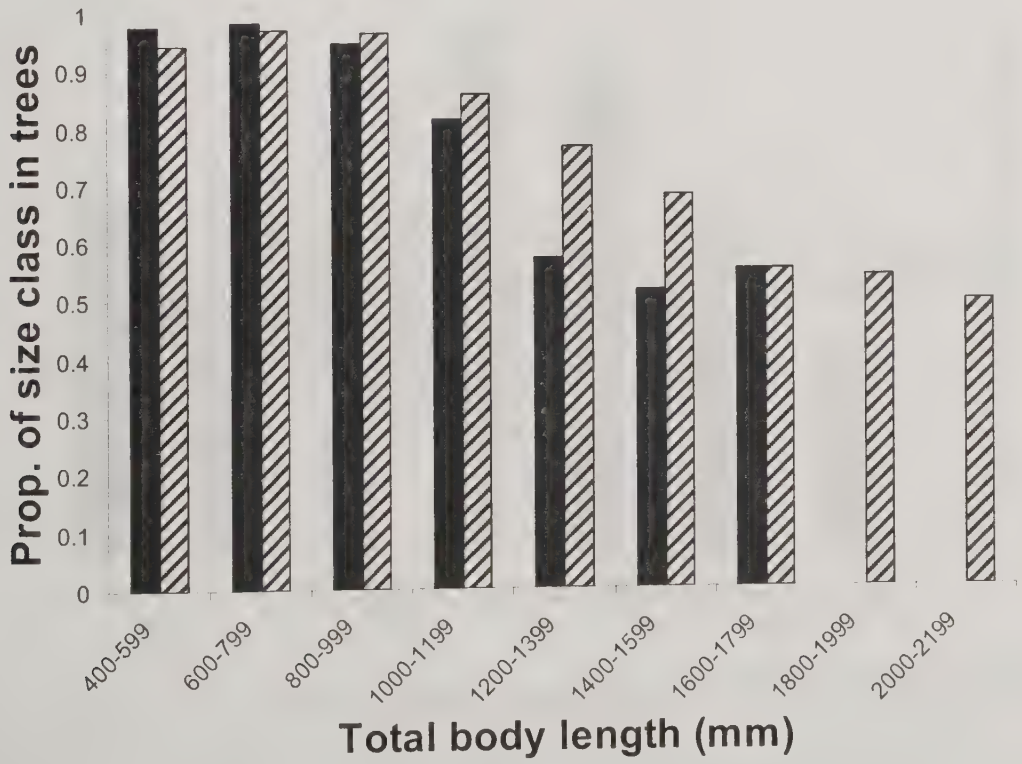



**Figure 2.** The proportion of BTS of various size classes that were encountered in trees during visual surveys on Guam. Black bars indicate data from the HIST dataset, while hatched bars indicate data from the NWFN dataset.

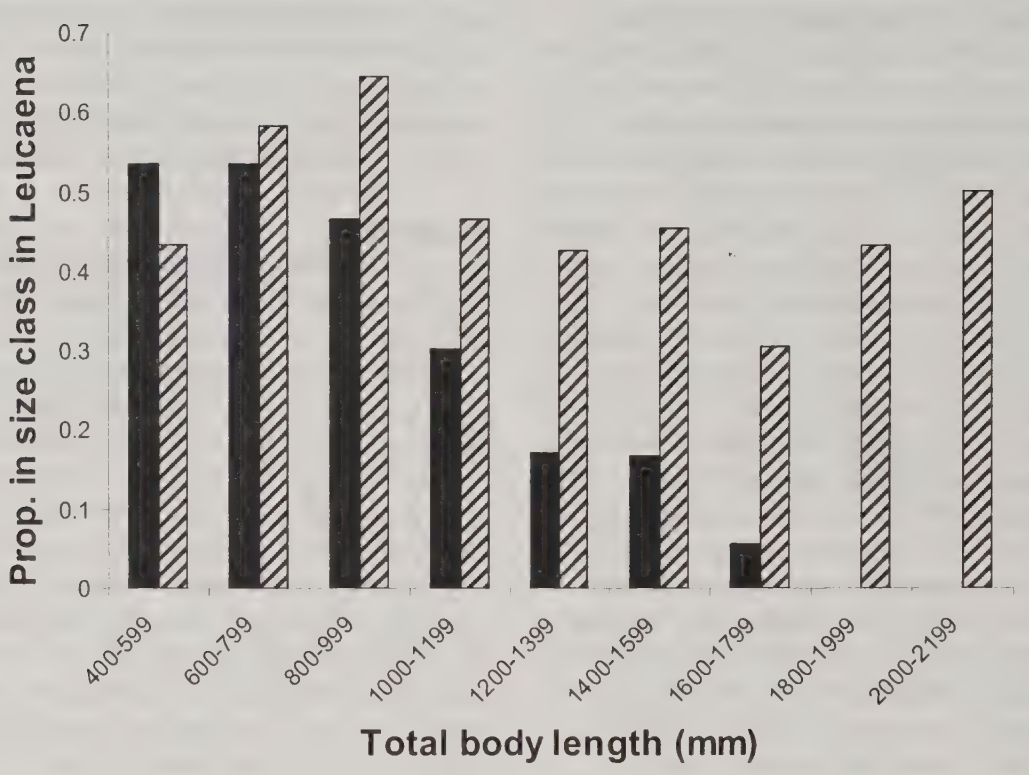



**Figure 3.** The proportion of BTS of various size classes that were encountered in tangantangan during visual surveys on Guam. Black bars indicate data from the HIST dataset, while hatched bars indicate data from the NWFN dataset.

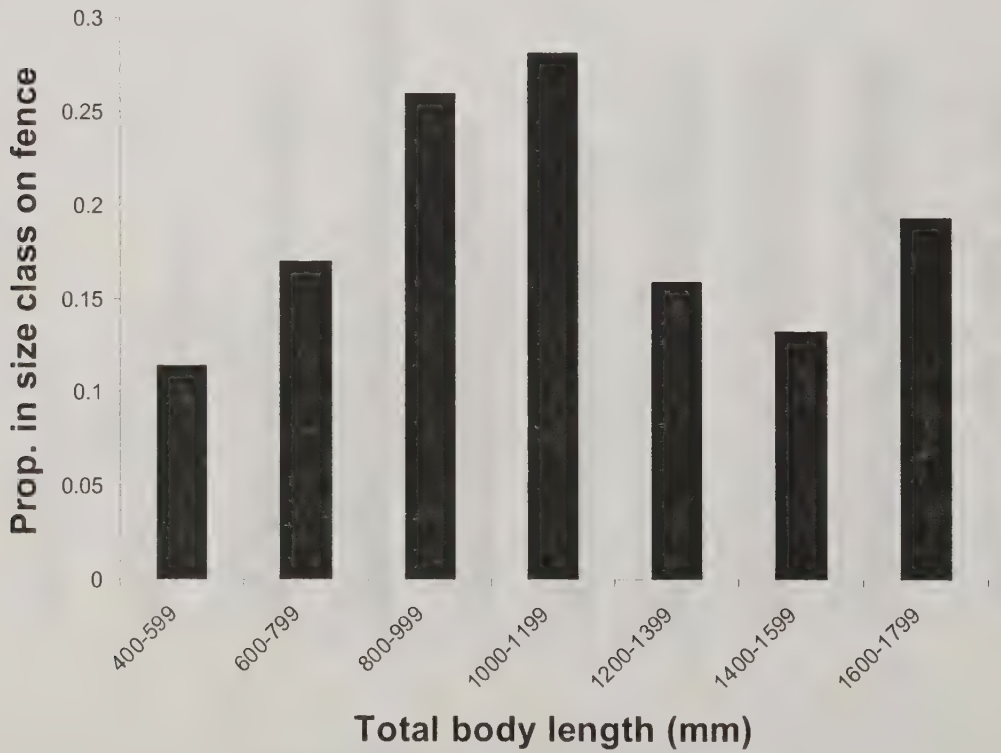



**Figure 4.** The proportion of BTS of various size classes that were encountered on fences during visual surveys on Guam. All data are from the HIST dataset.

Body length was a poor predictor of perch diameter, with variation in body size explaining <1% of variation in perch diameter (perch diameter $\log_{10}$-transformed, $F = 6.73$, $p = 0.01$, $r^2 = 0.0062$; solid diamonds in Figure 5). Among snakes in trees (i.e., those that were not on the ground and not on fences), total body length was also unrelated to vertical height of snakes when first observed (vertical height $\log_{10}$-transformed, $F = 3.04$, $p = 0.081$, $r^2 = 0.0027$; solid diamonds in Figure 6.)

Mean total body length of BTS in the NWFN dataset was 1,117 ± 373 mm and ranged from 405 to 2,240 mm; this dataset included 67 captures of BTS that were longer than the longest snake in the HIST dataset. As with the HIST dataset, >90% of small snakes were found in trees (hatched bars in Figure 2), and the mean length of BTS found on the ground was 1,399 ± 337, or ~24% larger than the overall average. Mean length of BTS in tangantangan was 1,069 ± 365 mm total length, ~4% smaller than the overall average (hatched bars in Figure 3); the tendency for decreased use of tangantangan use by large snakes was not as strong as for the HIST dataset. Because the NWFN dataset was largely from forested habitats with a boundary barrier that repelled climbing attempts by snakes, there were few observations of snakes on fences and thus no basis for comparison with HIST. Mean perch diameter of snakes in vegetation was 15.1 ± 26.4 mm and ranged from 1 to 500 mm. Similar to the results for the HIST dataset, total body length was related to perch diameter but explained <6% of variation in perch diameter (perch diameter $\log_{10}$-transformed, $F = 62.63$, $p < 0.0001$, $r^2 = 0.057$; hollow circles in Figure 5). Total body length was once again unrelated to vertical height of snakes when first observed in trees (vertical height $\log_{10}$- transformed, $F = 0.799$, $p = 0.371$, $r^2 = 0.0008$; hollow circles in Figure 6).

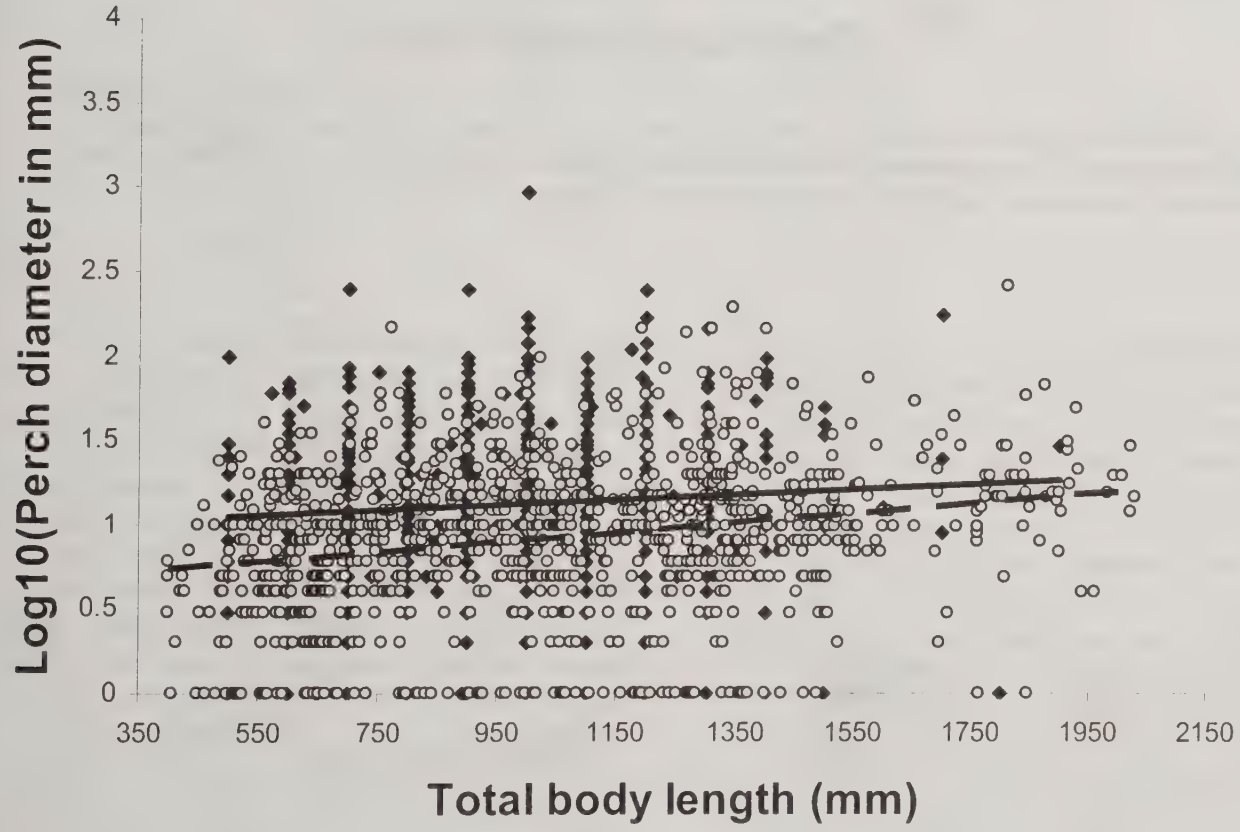



**Figure 5.** Perch diameter ($\log_{10}$-transformed) of BTS on Guam when first observed as a function of their total body length, with lines of best fit. Solid diamonds and solid line indicate data from the HIST dataset ($r^2 = 0.006$), hollow circles and dashed line indicate data from the NWFN dataset ($r^2 = 0.057$).

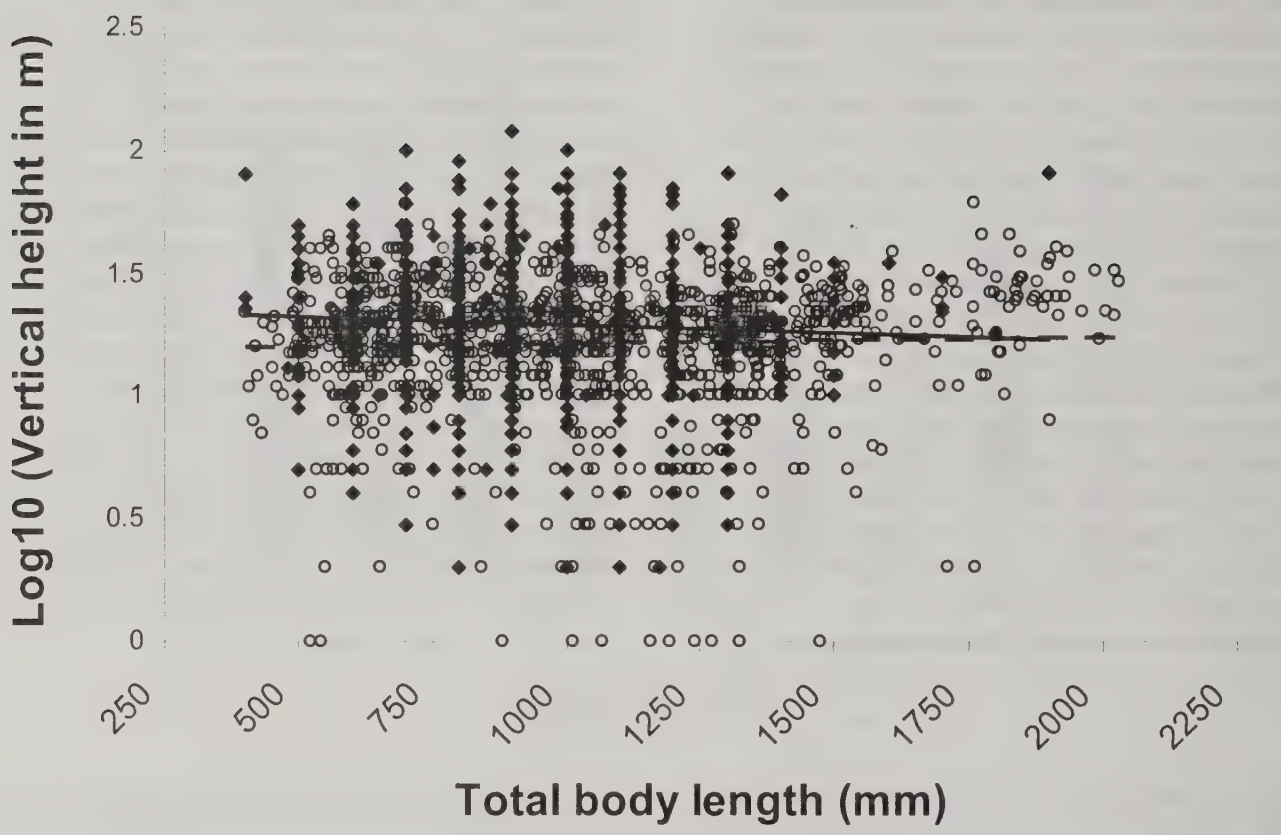
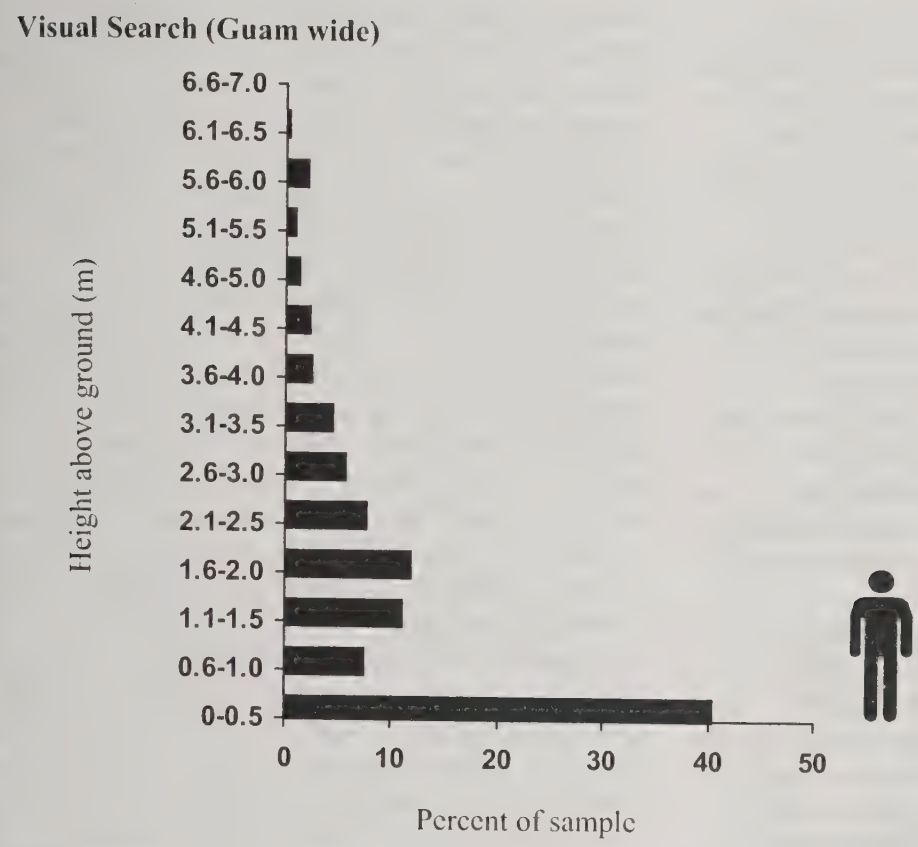



**Figure 6.** Vertical height ($log_{10}$-transformed) of BTS on Guam when first observed as a function of their total body length, with lines of best fit. Solid diamonds and solid line indicate data from the HIST dataset ($r^2 = 0.003$), hollow circles and dashed line indicate data from the NWFN dataset ($r^2 = 0.001$).

## DISCUSSION

Research on cryptic, nocturnal species such as BTS is often complicated by observer bias and poorly quantified detectability of different ages, sexes, or stages of the target organism. This is certainly true for BTS, as there is a great deal of inter-observer variability in the ability to find snakes via visual searches (Rodda and Fritts 1992) and rates of visual sightings are often only weakly correlated with abundance as determined by analysis of mark-recapture data (Rodda et al. 2005). However, we feel that many of the conclusions discussed below are robust to such sampling problems. For example, recent data suggest that nocturnal visual surveys are effective in detecting BTS of all body size classes (Rodda et al. 2007), such that our use of proportions, rather than absolute counts, of snakes of different sizes in different habitats is a conservative approach. Human observers may have a bias towards detecting snakes at eye level, as snakes from 1.5-2.25 m above ground are more likely to be observed during visual searches; this could potentially bias observations of vertical habitat use. We note that the overall distribution of perch heights used by snakes in the HIST dataset was qualitatively similar to the distribution of heights observed in free-ranging snakes by Clark (1998, Figure 7). In the latter study, Clark (1998) attached thread bobbins to free-ranging BTS on Guam upon capture and then immediately released them to their original position. The trailing threads were followed the day after release and heights of the thread were measured at one meter intervals. The fact that our datasets include large sample sizes of all body size classes at various heights suggests that our use of proportional abundances was unlikely to have been biased by lack of observations for some size classes.

Having assessed the potential pitfalls of analyses based on visual observations, what can we conclude about behavior and habitat use of small



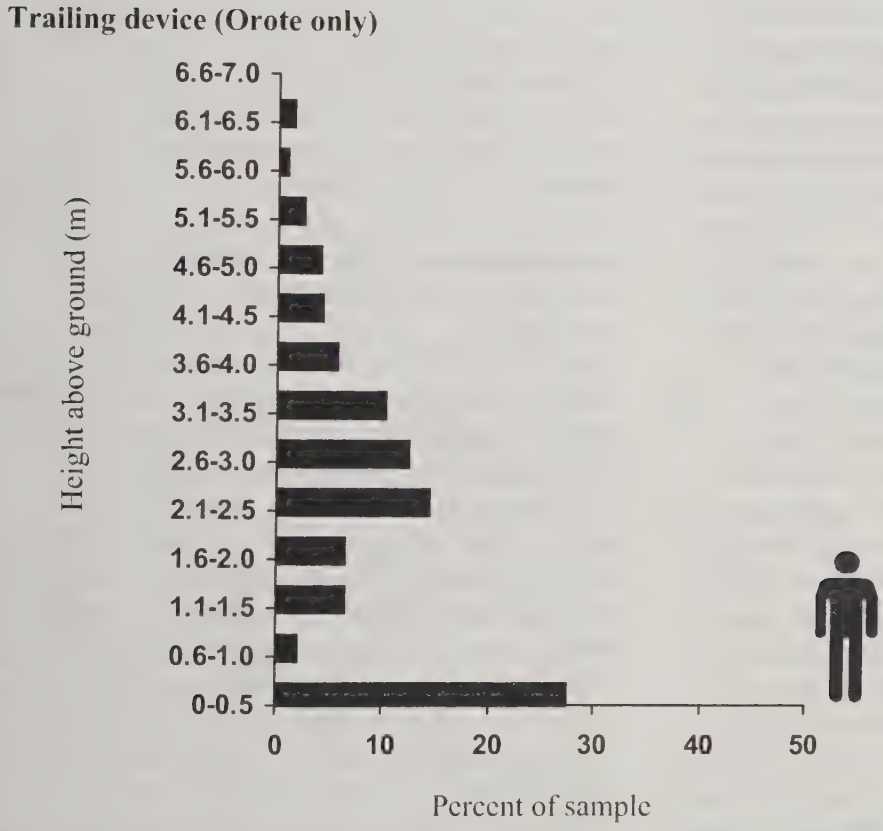




**Figure 7.** Comparison of observed heights of BTS when first observed in various habitats on Guam (Visual Search, from HIST dataset, on left) with vertical heights used by BTS that were released after being fitted with a thread-trailing device on Orote Peninsula, Guam (Trailing Device, on right, data courtesy C.S. Clark). Human figure added for scale.

BTS on Guam? The most striking trend was the difference between small and large snakes in their tendency to use arboreal habitats. Small snakes were overwhelmingly found in trees, while large snakes used trees and ground habitats in roughly equal proportions. Such results were highly concordant among the two datasets; snakes less than about one meter in length were almost always found in trees, but this proportion dropped to about 80% when snakes exceeded this size threshold, and decreased to roughly 50% at the largest size classes. More frequent use of terrestrial habitats by large snakes may be a foraging tactic that allows large snakes to more frequently encounter non-native small mammals (*Rattus*, *Mus*, and *Suncus*), or may represent movement for other reasons, e.g., mate-searching behavior.

Although the data indicated a difference between small and large snakes in their use of arboreal habitats, the vertical heights used by snakes were not related to snake body length. Instead, BTS appeared to use all available vegetative heights regardless of the length of the snake. Similarly, there was no clear pattern of perch diameter used by BTS as a function of body length. Small and large snakes were equally able to use perches of varying diameters, rather than being forced to use larger-diameter perches as body size increases. Brown treesnakes are extremely gracile (Rodda et al. 1999b), and even large snakes can use small-diameter perches by distributing their weight across several such perches.

Although Clark (1998) found a slight trend towards arboreal BTS using heights between 2 and 3.5 m, this may have been due to constraints on the available vegetative heights in forests on Guam, which is frequently battered by typhoons. The vertical microhabitats used by BTS from NWFN reflected the heights available to snakes; forests in the closed population were composed of young, early-successional trees that infrequently exceeded four meters in height during the sampling periods in the NWFN dataset, and this limitation was manifest in the fairly smooth upper limit to vertical heights of snakes in this dataset as compared to that observed in the HIST dataset from various forest types.

Taken together, these observations about arboreal habitat use by BTS have important management implications. Current trapping efforts typically include traps placed 1-2 m above ground level (Rodda et al. 1999a), facilitating trap-checking by humans. The apparent lack of a tendency for small BTS to prefer perches high in trees means that attempts to target such small snakes for control purposes need not include placing traps or bait stations high in trees, but that the standard operational trap heights should suffice. Logistical and safety issues associated with placing control devices high in trees might thus be avoidable for management purposes. However, we stress that this conclusion depends on the assumption that there is no inter-individual heterogeneity in the tendency to use certain perch heights. If, for example, some individuals consistently select high perches, then these individuals may avoid capture by traps placed at standard heights. Further research is required to address this question.

The HIST dataset indicates that tangantangan appeared to harbor high numbers of juvenile snakes. Detectability of BTS may be higher in tangantangan than in broad-leafed trees, as tangantangan leaflets fold up at night and observers can thus see deeper into the tree's canopy. However, observations of a high proportion of small snakes in this habitat did not appear to be an artifact of differential detectability across size classes, since medium- to large-sized snakes should have been even more visible in tangantangan. Instead, small snakes were likely keying in on the high prey densities in this habitat type. For example, Rodda et al. (2005) found that mourning geckos (*Lepidodactylus lugubris*) were present in densities >4,400/ha in some habitats on Guam, and that the density of this species can be even higher in tangantangan forests (Rodda unpublished data). These geckos are an appropriately-sized meal for juvenile BTS.

While the NWFN dataset suggested that small snakes are slightly more likely to use tangantangan than were large snakes, larger snakes also used this habitat to a greater extent than was observed in the HIST dataset. However, this was probably an artifact of the habitat available to snakes in the closed population. This area started as a scrubby plant community primarily composed of low (<1.5 m) vegetation. Erection of the snake barrier also excluded feral pigs (*Sus scrofa*) and deer (*Cervus mariannus*), allowing the vegetation inside to flourish. Tangantangan now comprises a large proportion of the forest inside the closed population. Use of this tree species by larger snakes was likely a function of its overwhelming prevalence, such that arboreal movements by large snakes were largely limited to tangantangan because there were few other choices. Habitat use by BTS in the closed population might be

considered a poor example of habitat choice, since snakes were confined to a 5-ha area with limited habitat diversity. For example, Santana-Bendix et al. (1994) used radiotelemetry to determine the activity ranges of free-ranging adult BTS on Guam, and found that mean net distance moved per night was 54.7 m. If adults in the closed population were moving at even a fraction of this nightly rate, then they may frequently have "bounced" off the barrier and back into tangantangan forests where they may be detected at a higher frequency than they would be in more heterogeneous forests outside the closed population barrier.

Differential use of tangantangan by BTS of various size classes offers insights for interdiction on Guam. Habitats composed of virtually monotypic tangantangan forest are common in disturbed areas on Guam, including many forests surrounding ports of exit. These forests may harbor high densities of juvenile snakes, which are more likely to bypass traps placed on fencelines or only forest perimeters for interdiction purposes, however, see Vice and Vice (2004). Low capture rates of snakes in these traps may not necessarily reflect low BTS densities in these areas, as traps generally fail to capture small snakes (Rodda et al. 2007). While visual searches of fencelines would seem a partial solution to the size bias of traps, this is contra-indicated by the very low capture rates of small snakes on fences. Some relief from this dilemma may be afforded by the apparent avoidance of terrestrial habitats by small snakes – maintenance of a buffer of cleared ground on each side of a fenceline will likely reduce the odds of small snakes moving from forests into cargo areas and flightlines.

Heavy use of tangantangan by small, but not large, BTS in natural habitats may offer insights for detection of incipient BTS populations on islands other than Guam, especially in the Northern Mariana Islands where tangantangan is abundant. Consider a situation where a large snake is reported by an observer on Saipan, resulting in a search deployment by local agency personnel or the USGS Rapid Response Team (Stanford and Rodda 2007). If the goal is to find such a large snake by implementing the results discussed above, then searchers should remember to spend roughly equal amounts of time searching arboreal and terrestrial habitats. However, it is important to remember that the purpose of a Rapid Response deployment is not expressly to find the individual snake reported by an observer (Stanford and Rodda 2007). Deployments (and intensive search efforts by local cooperating agencies) instead function to detect and delineate incipient populations of BTS. Incipient populations are those in which reproduction has started to occur. Because gravid BTS are rarely found and eggs are extremely difficult to find (Savidge et al. 2007), evidence of reproduction is best discovered by finding hatchling and juvenile snakes. These small individuals use tangantangan extensively and can be easily detected in tangantangan at night. Assuming that BTS habitat use in an incipient population is similar to that on Guam, incipient populations could perhaps be more easily identified by concentrating search efforts near the original snake sighting, but in tangantangan habitats.

The scarcity of very small snakes in fenceline captures, as revealed by the HIST dataset, indicates that while small snakes are typically found in arboreal habitats, not all vertical substrates are equal. This result seems puzzling, as nocturnal spotlighting of chain-link fences typically reveals large numbers of geckos (Rodda 1991) that would seem to be ideal prey for small snakes. One possible explanation is that smaller snakes are less likely to be seen by human observers during fenceline searches, thus biasing captures towards larger snakes. However, this seems unlikely as the color and reflectance of BTS makes them very obvious when illuminated on a fence and a fence can be completely scanned with relative ease. An alternative explanation is that most fences are separated from forested habitat by open ground. If small snakes are somewhat loath to cross open ground, as suggested by the results above, then these small individuals may be largely filtered out of the pool of animals available for capture on fences. The relative absence of very large snakes from fenceline captures may be due to their relative scarcity in the population's sampled or low densities of suitable-sized prey (e.g., rats, birds) on fences at night. If large prey items are uncommon, then fences may hold few odors from these prey types and, thus, may be chemically unattractive to large snakes. Further observations and behavioral trials are needed to assess these hypotheses.

Engeman and Vice (2001) examined sizes of BTS captured on fences via spotlight searches in November and December of 1998. Although exact sample sizes are not given in their paper, the number of snakes taken from fences appears to be ~54, based on stated monthly means. They observed that only 3.7% of these captures were of snakes <750mm SVL, but that 81.5% of their sample was <1,000 mm SVL (for comparison with

our paper, this is roughly equivalent to 1275 mm total body length). This trend is reinforced by the HIST dataset, which comprises roughly an order of magnitude more observations of snakes captured on fences (N = 568). Engeman and Vice (2001) used these results to conclude that "trapping was at least as effective for capturing small (sub-750 mm) snakes as was spotlighting fences." We note that capture rate of small snakes using either method is relatively low as calculated by Engeman and Vice (2001), and that more recent intensive research reveals that trapping is ineffective for small juveniles (<700 mm SVL, Rodda et al. 2007). Capture rates of small snakes on fencelines are, therefore, higher than rates from trap captures – however, these rates are low compared to mid-sized snakes, and extremely low in relation to their probable abundances in the population being sampled (Rodda et al. 2007).

## CONCLUSION

We conclude that long-term datasets of BTS captures via visual searches were largely concordant with intensive visual surveys of a geographically-closed population. These historical datasets, with large sample sizes from a variety of habitat types, may thus offer useful information when examining habitat-specific or temporal trends in various aspects of BTS behavior and habitat use. Our results indicated that small BTS were almost exclusively arboreal, while large snakes used arboreal and terrestrial habitats in roughly equal proportions. Juvenile BTS appeared to prefer to forage in tangantangan to a much greater extent than did adults, perhaps due to an abundance of small-bodied prey in this habitat type. Fenceline searches yielded very few small BTS, but fencelines may be used slightly more often by mid-sized snakes.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

CLARK, C. S. 1998. Activity patterns and microhabitat use of the brown treesnake, *Boiga irregularis*. Unpublished MS thesis, Ohio State University, Columbus, Ohio, USA.

ENGEMAN, R. M. AND D. S. VICE. 2001. A direct comparison of trapping and spotlight searches for capturing brown treesnakes on Guam. Pacific Conservation Biology 7:4-8.

RODDA, G. H. 1991. Fence climbing by the arboreal snake *Boiga irregularis*. The Snake 23:101-103.

RODDA, G. H., E. W. CAMPBELL, T. H. FRITTS, AND C. S. CLARK. 2005. The predictive power of visual searching. Herpetological Review 36:259-264.

RODDA, G. H. AND T. H. FRITTS. 1992. Sampling techniques for an arboreal snake, *Boiga irregularis*. Micronesica 25:23-40.

RODDA, G. H., T. H. FRITTS, C. S. CLARK, S. W. GOTTE, AND D. CHISZAR. 1999a. A state-of-the-art trap for the brown treesnake. Pages 44-80 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and brown treesnake. Cornell University Press, Ithaca, New York, USA.

RODDA, G. H., T. H. FRITTS, M. J. MCCOID, AND E. W. CAMPBELL. 1999b. An overview of the biology of the brown treesnake, a costly introduced pest on Pacific Islands. Pages 44-80 *in* G. H. Rodda, Y. Sawai, D. Chiszar, and H. Tanaka, editors. Problem snake management: the habu and brown treesnake. Cornell University Press, Ithaca, New York, USA.

RODDA, G. H. AND J. A. SAVIDGE. 2007. Biology and impacts of Pacific island invasive species. 2. *Boiga irregularis*, The brown treesnake (Reptilia: Colubridae). Pacific Science 61:307-324.

RODDA, G. H., J. A. SAVIDGE, C. L. TYRRELL, M. J. CHRISTY, AND A. R. ELLINGSON. 2007. Size bias in visual searching and trapping of brown treesnakes on Guam. Journal of Wildlife Management 71:656-661.

SANTANA-BENDIX, M., E. MAUGHAN, V. MERETSKY, AND C. B. SCHWALBE. 1994. Movement and activity patterns of *Boiga irregularis* (Colubridae) introduced predator on the island of Guam. Final report submitted to National Ecology Research Center, National Biological Survey, Fort Collins, Colorado, USA.

SAVIDGE, J. A. 1987. Extinction of an island forest avifauna by an introduced snake. Ecology 68:660-668.

SAVIDGE, J. A., F. J. QUALLS, AND G. H. RODDA. 2007. Reproductive biology of brown treesnakes, *Boiga irregularis*, during their colonization of Guam and in comparison to its native range. Pacific Science 61:187-195.

SHINE, R. AND X. BONNET. 2000. Snakes: a new 'model organism' in ecological research? Trends in Ecology and Evolution 15:221-222.

STANFORD, J. S. AND G. H. RODDA. 2007. The brown treesnake rapid response team. Pages 175-217 *in* G. W. Witmer, W. C. Pitt, and K. A. Fagerstone, editors. Proceedings of the managing vertebrate invasive species. August 2007. Fort Collins, Colorado, USA.

VICE, D. S., R. M. ENGEMAN, AND D. L. VICE. 2005. A comparison of three trap designs for capturing brown treesnakes on Guam. Wildlife Research 32:355-359.

VICE, D. S., AND D. L. VICE. 2004. Characteristics of Brown Treesnakes removed from Guam's transportation network. Pacific Conservation Biology 10:216-221.

# EARLY DETECTION AND ERADICATION OF INVADING RATS

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***Abstract:*** Invasive rats continue to colonize rat-free islands around the world. To prevent rats from establishing on rat-free islands, especially following their eradication, biosecurity actions are required to enable early detection and elimination. Rats arrive at islands by both human transportation and by swimming. There are very little data on the rates of rat transportation by humans, although it is known that they are not negligible. There are better data on the distances rats can swim, allowing estimates to be made of the risk of reinvasion of islands close to source populations. Biosecurity prioritization must take place across all rat-free islands, balancing the likelihood and impact of rat establishment. Dense grids of poison bait stations are not preferable for preventing rat invasion. Instead, surveillance systems that integrate multiple device types appear to be best for intercepting invading rats, but must be tested to ensure they are effective. This can be done by releasing a controlled number of monitored rats onto a rat-free island. Islands can now be maintained rat-free despite non-negligible reinvasion rates; however, in some cases islands must be managed within a larger meta-population context and eradication may never be achieved.

***Key Words:*** biosecurity, detection, invasion, invasive species, islands, *Rattus*, risk, rodents.

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## INTRODUCTION

Invasive species continue to colonize new locations, facilitating the homogenization of the world's biodiversity on a global scale (Collins et al. 2002). Islands are particularly vulnerable to the changes caused by non-native species (Courchamp et al. 2003). Invasions by non-native species should not be considered inevitable. Preventative measures (biosecurity) can be effective, especially when there are natural barriers to colonization. Biosecurity is most easily implemented on islands, where the natural water barrier acts as a filter the arrival of species, and arrivals are often focused at points of activity (e.g., ports/airstrips). Effective biosecurity that prevents invasion is economically more cost-effective than responding after an invasion has happened (Leung et al. 2002).

Three invasive rat species (*Rattus exulans*, *R. norvegicus* and *R. rattus*) have collectively colonized every continent except Antarctica, and over 80% of the world's oceanic island groups (Atkinson 1985). Invasive rats impact on human health, and agricultural, economical and conservation values (Singleton et al. 2003). Their negative impacts on insular ecosystems and species are becoming increasingly well documented (Towns et al. 2006, Jones et al. In Press), and they continue to invade historically rat-free islands at the same rate over the last century (Russell et al. In Press-b). As the emphasis on permanent rodent removal (eradication) from islands increases (Howald et al. 2007), there may be a concurrent increase in the rate of rat reinvasion to islands that have recently become rat-free.

To continue keeping rats permanently off rat-free islands, there will need to be an increased emphasis on island biosecurity. In this paper, we outline risk assessment for island invasion by rats, and biosecurity methods that would minimize the risk of invasive rat establishment on rat-free islands. We also describe recent experimental trials where rats were released onto rat-free islands to test biosecurity systems.

## RISK ASSESSMENT

Risk assessment considers the likelihood and potential impact of an invasive species establishing in a new location (Andersen et al. 2004). Establishment of invasive species is usually considered along a pathway, which consists of propagule uptake, transport, release, survival and spread (Colautti and MacIsaac 2004). The further along this pathway invasive rats survive, the greater the likelihood of adverse impacts. Unfortunately, there are currently few data available to quantify the relative survival rates of invasive rats through

each of these stages. Considering the island as the unit of conservation, pre-border risk assessment considers propagule uptake, transport and release, while post-border efforts consider release, survival and spread.

Invasive rats are transported by human vessels and by natural locomotion (i.e., swimming). All long-distance movements of rats are controlled by uptake, transport and release from ocean-going vessels. Pre-border biosecurity efforts (e.g., by national customs agencies) are required to prevent invasive rats from arriving in new locations via this route. International Health Regulations (2005) require vessels to hold either a current Ship Sanitation Control Certificate or a Ship Sanitation Control Exemption Certificate, which have a life-time of six months. These certificates are issued on behalf of the World Health Organization in order to prevent the spread of contagions, and incorporate the previous (1969) De-rat Certification requirement. This certification provides obvious conservation benefits. Any ship deemed infested by rats shall be fumigated to a standard eligible for certification. Unfortunately, once a ship has cleared quarantine, it is free to move around within national waters, during which time it may transport rats within a region (e.g., from a mainland port to rat-free offshore islands). This kind of rat transportation is beyond the scope of international custom laws. Furthermore, vessels engaged in activities such as illegal fishing within foreign territorial limits increase the risk of unchecked rat transportation. Few data are currently available on the rate of ship infestation by rats. In Alaska, single rats (but not breeding populations) have been detected on large fishing vessels. In the 2006-2007 financial year, 0.3% (2 ships and 8 containers) of 3,119 international vessels entering New Zealand had rats or rat sign (A.Baker, personal communication), while less than 0.2% of 14,200 international vessels inspected entering Australia had live rodents or signs of recent infestation (D. Franks, personal communication). In 1999, ship rats arrived at Clipperton Island off the Baja Peninsula (Pitman et al. 2005), and in 2000, a Korean fishing vessel grounded on rat-free McKean Island, releasing Asian ship rats (*R. tanezumi*) onto the island (M. Thorsen, personal communication).

For islands very close to shore (1-2 km), invasive rat transport can occur by local transportation on vessels or by swimming. It can often be difficult to distinguish the mechanisms by which invasive rats are arriving on islands, although population genetics can provide tools to determine the points of departure (Abdelkrim et al. 2007). Very few data exist on the rates of vessel infestation for local transportation, although in southern New Zealand single rats have been recorded on a number of small vessels in recent years (Russell et al. In Press-b). In New Zealand, better data are available on rat swimming rates, which suggest that rats may swim further and more often than currently believed (Russell and Clout 2005). Invasive rats, particularly brown rats (*R. norvegicus*), can be very capable swimmers (e.g., Russell et al. 2005). Rates of rat arrival at islands can often only be accurately determined following rat eradication. These local movements by invasive rats can be considered post-border in that vessels have cleared international customs, although they remain pre-border in an island biosecurity context, since rats have not yet arrived on the island unit of conservation interest.

Risk assessment for invasive rats on islands balances the likelihood of rat establishment with the potential severity of rat impacts. Rat-vulnerable species often survive, or are 'marooned', on islands (Hutton et al. 2007), contributing to the conservation values of those islands. The application of risk management actions (i.e., biosecurity) essentially becomes a prioritization activity across all rat-free islands, and requires good data on invasive rat presences, transportation likelihoods and the presence of vulnerable species. In New Zealand, biosecurity capacity for invasive rats (as well as other species) has gradually been increased as more islands have become rat-free and the long-term benefits of rat eradication have been observed. This has also essentially made island biosecurity very site-specific. Only recently has a nationwide approach to island classification been adopted by the Department of Conservation in New Zealand (Anonymous 2007).

## BIOSECURITY

When a single rat arrives on an island, it is known as an incursion; if other individuals arrive and a population is established, it is an invasion. Biosecurity measures can be implemented as a form of risk management (Andersen et al. 2004), comprising quarantine, surveillance and contingency response. Quarantine involves minimizing the possibility of rat transport during and following arrival on an island. Surveillance includes the actions taken to monitor for rats both on and off islands (e.g., on boats or at points of departure), and requires a long-term, continuous

commitment, so that incursions are reliably detected and responded to earlier and hence more successfully. Once there is a quarantine breach or surveillance detection, a contingency response is launched (e.g., Wace 1986). A contingency response is a calculated, and usually expensive and laborious, exercise to eliminate the invading rats using a combination of methods (e.g., Russell et al. 2005). It must be made immediately and with the same intensity as an eradication campaign. Before an incursion actually occurs it is important that the capacity and planning for a contingency response are well in place.

Biosecurity to prevent rat invasion of islands could be prioritized by island invasion rate, economic costs of biosecurity, or costs to biodiversity. We provide an indication of the earliest points where managers may wish to consider biosecurity intervention given different prioritization factors and levels (Figure 1). Biosecurity to prevent rat reinvasion is an on-going and intensive process (Witmer et al. 2007).

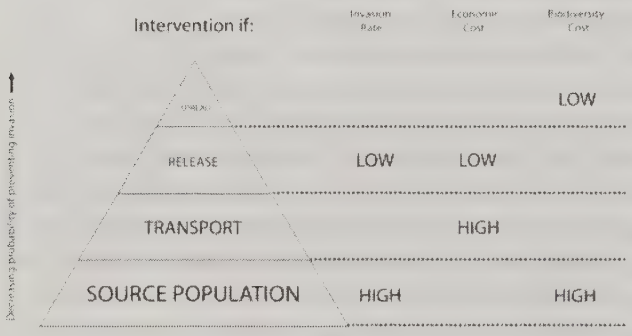



**Figure 1.** The likelihood of preventing rat establishment decreases the further along the invasion pathway intervention occurs. Earliest points of intervention could be prioritized by levels of invasion rates, economic costs or biodiversity costs.

Two recent reviews are relevant to island biosecurity for invasive rats. Clapperton (2006) reviewed the behaviour of rodents in relation to control devices and emphasized the important differences between the behaviours of the three invasive rat species, which must be considered during biosecurity. Russell et al. (In Press-b) reviewed tools for preventing rat invasion of islands. Rodent behaviour interacts with control devices, which will affect how island biosecurity is undertaken (O'Connor and Eason 2000). In particular, methods that have previously been successful for eradicating rats at high densities may not be appropriate for biosecurity at low densities (Russell et al. 2005). Dense grids of bait stations may be particularly ineffective, as they can evoke neophobia (Russell et al. 2005), be inappropriately designed for rats (Spurr et al. 2006, Spurr et al. 2007) and introduce large volumes of poison to an ecosystem (Hoare and Hare 2006). Additionally, the success of the operation must often be inferred from poison take rather than corpse recovery. Since invading rats usually roam widely (Russell 2007), entry rates of devices are probably more important than encounter rates. Further work is required to develop methods for reliably intercepting invading rats in the presence of other small mammal species.

To test the success of biosecurity systems when there are low densities of invading rats, Russell et al. (In Press-a) released single invading brown rats onto various rat-free islands around New Zealand. A permanent surveillance system of multiple device types was found to be most successful at intercepting invading rats. It took more than 2 weeks to intercept some rats, where it was necessary to launch a contingency response using alternative methods. Complete island coverage enabled close to 100% interception. However, on larger islands where complete coverage is not possible, some rats could evade detection for many months. In the rat-free Pribilof Islands of Alaska, biosecurity around the two main ports consists of traps and poison, and has so far successfully prevented rat establishment despite a number of incursions (Sowls and Byrd 2002). The absence of rat detection does not guarantee that a biosecurity system is successfully preventing rat arrival, however; it may just mean the invasion rate is very low. Releasing a known number of rats onto a rat-free island is a powerful method for testing island biosecurity systems, and could be undertaken following rat eradication before the reintroduction of threatened species. Further work is still required to develop methods for confirming the successful removal of every individual (Kean and Suckling 2005, Morrison et al. 2007).

## CONCLUSION

The ability to successfully remove invasive rats from very large islands (up to 11,330 ha) increases the need for effective biosecurity to ensure that reinvasions are prevented. Recent research and management has shown that is it possible to intercept most invading rats within a few weeks of

arrival, provided that a suitable biosecurity system is in place. Even islands with relatively high invasion rates (a few rats per year) can be maintained rat-free. Biosecurity is an ongoing intensive exercise requiring the ability to rapidly respond to a suspected incursion with the same intensity as an eradication campaign. For islands that are very close to a source population, invasion rates eventually become so high that even widespread biosecurity cannot prevent rat establishment (e.g., Russell et al. 2007). In this case, islands must be effectively managed as part of a larger meta-population, where ongoing control is required and it is accepted that total 'eradication' cannot be definitively achieved.

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## LITERATURE CITED

ABDELKRIM, J., M. PASCAL, AND S. SAMADI. 2007. Establishing causes of eradication failure based on genetics: case study of ship rat eradication in Ste. Anne archipelago. Conservation Biology 21:719-730.

ANDERSEN, M. C., H. ADAMS, B. HOPE, AND M. POWELL. 2004. Risk assessment for invasive species. Risk Analysis 24:787-793.

ANONYMOUS. 2007. Draft offshore and outlying island strategy. Department of Conservation, Christchurch.

ATKINSON, I. A. E. 1985. The spread of commensal species of *Rattus* to oceanic islands and their effects on island avifaunas. Pages 35-81 *in* P.J. Moors, editor. Conservation of Island Birds. International Council for Bird Preservation Technical Publication No. 3, Cambridge, United Kingdom.

CLAPPERTON, B. K. 2006. A review of the current knowledge of rodent behaviour in relation to control devices. Science for Conservation 263. Department of Conservation, Wellington, New Zealand.

COLAUTTI, R. I., AND H. J. MACISAAC. 2004. A neutral terminology to define 'invasive' species. Diversity and Distributions 10:135–141.

COLLINS, M. D., D. P. VÁZQUEZ, AND N. J. SANDERS. 2002. Species–area curves, homogenization and the loss of global diversity. Evolutionary Ecology Research 4:457–464.

COURCHAMP, F., J. L. CHAPUIS, AND M. PASCAL. 2003. Mammal invaders on islands: impact, control and control impact. Biological Reviews 78:347-383.

HOARE, J. M., AND K. M. HARE. 2006. The impact of brodifacoum on non-target wildlife: gaps in knowledge. New Zealand Journal of Ecology 30:157-167.

HOWALD, G. R., C. J. DONLAN, J. P. GALVÁN, J. C. RUSSELL, J. P. PARKES, A. SAMANIEGO, Y. WANG, C. R. VEITCH, P. GENOVESI, M. PASCAL, A. SAUNDERS, AND B. TERSHY. 2007. Invasive rodent eradication on islands. Conservation Biology 21:1258-1268.

HUTTON, I., J. P. PARKES, AND A. R. E. SINCLAIR. 2007. Reassembling island ecosystems: the case of Lord Howe Island. Animal Conservation 10:22-29.

JONES, H. P., B. R. TERSHY, E. S. ZAVALETA, D. A. CROLL, B. S. KEITT, M. Y. FINKELSTEIN, AND G. R. HOWALD. In press. Global review of the severity of invasive rat impacts on susceptible seabirds. Conservation Biology.

KEAN, J. M., AND D. M. SUCKLING. 2005. Estimating the probability of eradication of painted apple moth from Auckland. New Zealand Plant Protection 58:7-11.

LEUNG, B., D. M. LODGE, D. FINNOFF, J. F. SHOGREN, M. A. LEWIS, AND G. LAMBERTI. 2002. An ounce of prevention or a pound of cure: bioeconomic risk analysis of invasive species. Proceedings of the Royal Society of London B 269:2407-2413.

MORRISON, S. A., N. MACDONALD, K. WALKER, L. LOZIER, AND M. R. SHAW. 2007. Facing the dilemma at eradication's end: uncertainty of absence and the Lazarus effect. Frontiers in Ecology and Environment 5:271-276.

O'CONNOR, C. E., AND C. T. EASON. 2000. Rodent baits and delivery systems for island protection. Science for Conservation 150. Department of Conservation, Wellington, New Zealand.

PITMAN, R. L., L. T. BALLANCE, AND C. BOST. 2005. Clipperton Island: pig sty, rat hole and booby prize. Marine Ornithology 33:193-194.

RUSSELL, J. C. 2007. Invasion ecology and genetics of Norway rats on New Zealand islands Unpublished PhD thesis. University of Auckland, Auckland.

RUSSELL, J. C., B. Z. BEAVEN, J. W. B. MACKAY, D. R. TOWNS, AND M. N. CLOUT. In press-a. Testing island biosecurity systems for invasive rats. Wildlife Research.

RUSSELL, J. C., AND M. N. CLOUT. 2005. Rodent incursions on New Zealand islands. Pages 324-330 *in* J. Parkes, M. Statham, and G. Edwards, editors. Proceedings of the 13th Australasian Vertebrate Pest Conference. Landcare Research, Lincoln.

RUSSELL, J. C., S. D. MILLER, H. E. MACINNES, AND R. M. FEWSTER. 2007. Genetic fingerprinting of rats from Pearl Island. Unpublished report, University of Auckland, Auckland, New Zealand.

RUSSELL, J. C., D. R. TOWNS, S. H. ANDERSON, AND M. N. CLOUT. 2005. Intercepting the first rat ashore. Nature 437:1107.

RUSSELL, J. C., D. R. TOWNS, AND M. N. CLOUT. In press-b. Review of invasion biology of rats: implications for island biosecurity. Science for Conservation, Department of Conservation, Wellington, New Zealand.

SINGLETON, G. R., L. A. HINDS, C. J. KREBS, AND D. M. SPRATT. 2003. Rats, mice and people: rodent biology and management. ACIAR monograph no. 96, Canberra, Australia.

SOWLS, A. L., AND G. V. BYRD. 2002. Preventing rat introductions to the Pribilof Islands, Alaska, USA. Page 413 *in* C. R. Veitch, and M.N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group, IUCN, Gland, Switzerland and Cambridge, United Kingdom.

SPURR, E. B., G. A. MORRISS, J. TURNER, C. E. O'CONNOR, AND P. FISHER. 2007. Bait station preferences of ship rats. DOC Research & Development Series 271. Department of Conservation, Wellington, New Zealand.

SPURR, E. B., C. E. O'CONNOR, G. A. MORRISS, AND J. TURNER. 2006. Bait station preferences of Norway rats. DOC Research & Development Series 255. Department of Conservation, Wellington, New Zealand.

TOWNS, D. R., I. A. E. ATKINSON, AND C. H. DAUGHERTY. 2006. Have the harmful effects of introduced rats on islands been exaggerated? Biological Invasions 8:863-891.

WACE, N. M. 1986. The rat problem on oceanic islands - research is needed. Oryx 20:79-86.

WITMER, G. W., F. BOYD, AND Z. HILLIS-STARR. 2007. The successful eradication of introduced roof rats (*Rattus rattus*) from Buck Island using diphacinone, followed by an irruption of house mice (*Mus musculus*). Wildlife Research 34:108-115.

# TREE SQUIRRELS AS INVASIVE SPECIES: CONSERVATION AND MANAGEMENT IMPLICATIONS

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***Abstract:*** The impact of invasive species on native biodiversity is increasing worldwide. Mammalian invaders are formidable ecological and economic threats, and tree squirrels are models of such challenges. We review the worldwide distribution of tree squirrel introductions and detail their nearly universal success. The biological characteristics of tree squirrels that enable success as invasive species are: high reproductive potential, high vagility, diverse food habits, ability to construct nests, and plasticity in human-impacted landscapes. We document costs and benefits of tree squirrel introductions, and discuss existing management strategies planned for some species. We focus on an introduced population of Mexican red-bellied squirrels (*Sciurus aureogaster*) found in Biscayne National Park, Florida, United States. Originally introduced to Elliot Key in 1938, the population expanded until Hurricane Andrew in 1992. Thought to be extirpated, recent sightings initiated a project to assess status and distribution within the Park. A field study has provided valuable information on potential and realized impacts that the squirrel population has on the native ecosystem that includes several species of conservation concern. Management strategies for Mexican red-bellied squirrels can be formed after research to develop prudent and effective means to meet the challenges that result from control measures to deal with this exotic species.

***Key Words:*** *Callosciurus*, exotic, Florida, *Funambulus*, invasive species, *Sciurus*, *Tamiasciurus*.

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## INTRODUCTION

Introductions of mammals outside of their native range have been commonplace often with significant negative impacts (Long 2003). Nine species of tree squirrel have been successfully introduced outside their native range, with nearly 100 individual cases of introduction (Long 2003). Here we include all species of the family Sciuridae that maintain an arboreal lifestyle, and do not limit our discussion strictly to the tribe Sciurini (Wilson and Reeder 1993). Tree squirrels have been introduced mostly for the aesthetic novelty and pleasure they bring (Aprile and Chicco 1999), but also to increase hunting and trapping opportunities (Davis and Brown 1988, Long 2003). We first review the major introductions of tree squirrels (Table 1), and then discuss the biology of tree squirrels that make them successful invasives. Finally, we discuss management and monitoring techniques for introduced populations of tree squirrels.

## SPECIES ACCOUNTS

### Eastern Gray Squirrel (*Sciurus carolinensis*)

The native range of the eastern gray squirrel includes the eastern United States (US) from the Gulf Coast to southern Canada (Hall 1981, Koprowski 1994a, Long 2003). *S. carolinensis* has been the subject of the most numerous and widest geographic range of tree squirrels introductions (Long 2003). The species has been introduced beyond its native range in eastern Canada and the US to localities within the United States and Canada (Koprowski 1994a, Long 2003). Introduced populations of *S. carolinensis* can be found in the Canadian provinces of southern British Columbia, southern Quebec, southern New Brunswick, Saskatchewan, and Nova Scotia (Banfield 1974, Peterson 1966). Numerous exotic

**Table 1.** Date and locations of tree squirrel (Family Sciuridae) introductions worldwide.

| Species Name | Date | State/Province/Region | Country | Reference |
|---|---|---|---|---|
| *Sciurus aberti* | 1940s, 1970s | Arizona | US | Davis and Brown 1988 |
| | 1929,40,55,69 | New Mexico | US | Davis and Brown 1988 |
| | 1898-1908 | Utah | US | Davis and Brown 1988 |
| *Sciurus aureogaster* | 1938 | Florida | US | Brown and McGuire 1969 |
| *Sciurus carolinensis* | circa 1900,37 | Victoria, Melbourne | Australia | Seebeck 1995 |
| | 1914 | numerous sites | Canada | Banfield 1974 |
| | 1876-1929 | numerous sites | U.K. | Long 2003 |
| | 1913 | | Ireland | Long 2003 |
| | 1948,66,94 | numerous sites | Italy | Bertolino and Genovesi 2003 |
| | 1958 | Baja Norte | Mexico | Huey 1964 |
| | 1890s | Cape Town | S. Africa | Macdonald et al. 1988 |
| | circa 1850's | California | US | Jameson and Peeters 1988 |
| | 1943 | Hawaii | US | Kramer 1971 |
| | not reported | Idaho | US | Larrison and Johnson 1981 |
| | 1919 | Oregon | US | Verts and Carraway 1998 |
| | 1958,66,68 | Montana | US | Hoffman et al. 1969 |
| | 1904-1952 | North Dakota | US | Hibbard 1956 |
| | not reported | Texas | US | Schmidly 2004 |
| | 1934,49,66,68 | Wisconsin | US | Long 2003 |
| | circa 1920 | Washington | US | Flahaut 1941 |
| *Sciurus niger* | 1890 | Ontario | Canada | Peterson 1966 |
| | early 1900s | California | US | Jameson and Peeters 1988 |
| | early 1900s | Colorado | US | Fitzgerald et al. 1994 |
| | not reported | Idaho | US | Larrison and Johnson 1981 |
| | not reported | Montana | US | Hoffman et al. 1969 |
| | prior to 1958 | New Mexico | US | Frey and Cambell 1997 |
| | 1935,38,41,53,54 | North Dakota | US | Hibbard 1956 |
| | not reported | Oregon | US | Verts and Carraway 1998 |
| | not reported | Texas | US | Frey and Cambell 1997 |
| | 1915 | Washington | US | Yocum 1950 |
| *Sciurus vulgaris* | not reported | | Germany | Long 2003 |
| | not reported | Caucasus | Crimea | Long 2003 |
| | not reported | | Kazakhstan | Long 2003 |
| | not reported | | Kyrgyzstan | Long 2003 |
| *Callosciurus erythraeus* | 1970 | Buenos Aires | Argentina | Aprile and Chicco 1999 |
| | 1970s | Cap d'Antibes | France | Gurnell and Wauters 1999 |
| | 1935,54 | | Japan | Setoguchi 1990 |
| *Callosciurus finlaysoni* | 1981 | | Itlay | Bertolino et al. 1999 |
| *Funambulus pennantii* | 1898 | numerous sites | Australia | Long 2003 |
| *Tamiasciurus hudsonicus* | 1963,64 | numerous sites | Canada | Payne 1976 |
| | 1920s | Alaska | US | Long 2003 |

populations can be found in urban areas of California within city parks and other wooded areas (Jameson and Peeters 1988). Introduced populations also occur in Lewiston and Moscow, Idaho (Larrison and Johnson 1981). Records of introduced individuals were collected beginning in the mid 1960s in Montana (Hoffman et al. 1969). Multiple introductions have taken place in North Dakota, and all have been successful in establishing breeding populations (Hibbard 1956).

Approximately 20 squirrels were transplanted from Pennsylvania to Marion County, Oregon, but the population has remained restricted to urban areas within the county while also being introduced to Sweet Home, Oregon (Verts and Carraway 1998). Although native to eastern Texas, *S. carolinensis* has been introduced into the western side of the state (Schmidly 2004). *S. carolinensis* has also been introduced to Fort Lewis, Seattle, and other urban areas in Washington (Flyger and Gates 1982)

where their range overlaps that of the state threatened western gray squirrel (*Sciurus griseus*, Washington Department of Wildlife 1993, Ryan and Carey 1995, Bayrakci et al. 2001). Wisconsin also received multiple introductions of this species (Long 2003). Two populations were established in northern Baja California, Mexico (Huey 1964); however, the fate of these populations remains unclear (Koprowski et al. 2006).

*S. carolinensis* has been introduced outside of North America to the British Isles, Italy, South Africa, Hawaii, and Australia (Koprowski 1994a, Long 2003). However, the Hawaiian population now appears extirpated (Kramer 1971), and the Australian populations in Melbourne and Ballarat have gone extinct (Lever 1985, Seebeck 1995, Long 2003). Introduced to Cape Town, South Africa, the species has spread to inhabit the Cape Peninsula including the Cape of Good Hope Nature Reserve (Haagner 1920, Lever 1985, Macdonald et al. 1988). Introductions as early as the 1820s allowed the species to establish on the British Isles, however multiple introductions occurring in the early 1900s allowed the squirrel to become a nuisance animal as it spread to occupy much of the former range of the native Eurasian red squirrel (*Sciurus vulgaris*, Lever 1985). *S. carolinensis* has been introduced to Italy on four separate occasions, with three populations being established (Long 2003, Bertolino and Genovesi 2005).

Numerous negative impacts have been reported from areas of introduction including damage to trees and agriculture (Mountford 1997, Rayden and Savill 2004, Mountford 2006), competition with native squirrels (Macdonald 1988, Washington Department of Wildlife 1993, Wauters et al. 2000, Bayrakci et al. 2001, Gurnell et al. 2004b), and transmission of disease to native species (Rushton et al. 2000, McInnes 2006). Gardens and orchards in South Africa were reported as damaged by introduced squirrels (Haagner 1920). *S. carolinensis* damages orchard, garden, and farm crops in the British Isles and is an ecological pest taking native bird's eggs and nestlings (Lever 1985). *S. carolinensis* also debarks trees and can severely affect the forest structure (Mountford 1997, Rayden and Savill 2004, Mountford 2006), however, impacts in some areas are minimal (Signorile and Evans 2007). Interspecific competition with native sciurids has occurred in many areas where *S. carolinensis* was introduced. Eastern gray squirrels now occur in areas of Washington occupied by the state threatened western gray squirrel (Washington Department of Wildlife 1993, Bayrakci et al. 2001), where the invasive squirrel is better able to acclimate to human development (Ingles 1947, Ryan and Carey 1995). *S. carolinensis* is more commonly observed in line transect studies in Fort Lewis than native western gray squirrels (Ryan and Carey 1995). *S. carolinensis* negatively impacts the native Eurasian red squirrel populations in the British Isles and Italy through competitive exclusion for space and food resources (Wauters et al. 2000, Bryce et al 2001, Gurnell et al. 2004b) and spreading a poxvirus to which *S. vulgaris* is highly susceptible, but which does not effect *S. carolinensis* (Rushton et al. 2000, McInnes 2006).

**Abert's Squirrel (*Sciurus aberti*)**

The Abert's squirrel is native to the pine forests of Arizona, Colorado, New Mexico, Utah, and Wyoming in the US as well as occurring in Sonora and Chihuahua into southern Durango in Mexico (Wilson and Reeder 1993). *Sciurus aberti* was transplanted outside its native range beginning in the early 1900s by different state game agencies to increase hunting opportunities, with most of these transplants occurring in the state of Arizona (Davis and Brown 1988). Beginning in the 1940s, Arizona Game and Fish Department released individuals from northern Arizona into many of the "sky island" mountain ranges in southeastern Arizona (Davis and Brown 1988). The only population to receive much scientific attention is located in the Pinaleno Mountains of southeastern Arizona, which is home to the endangered endemic Mount Graham red squirrel (*Tamiasciurus hudsonicus grahamensis*, United States Fish and Wildlife Service 1993, Hutton et al. 2003, Edelman and Koprowski 2005, Edelman and Koprowski 2006, Rushton et al 2006).

Abert's squirrels and Mount Graham red squirrels feed on many of the same foods and seek similar trees for nest sites (Hutton et al. 2003, Edelman and Koprowski 2005, Edelman and Koprowski 2006). *S. aberti* raid red squirrel middens and are chased by the territorial red squirrel (Hutton et al. 2003). All of these factors are of conservation concern for the Mount Graham red squirrel, as modeling reveals that the continued presence of the Abert's squirrel could severely impact the viability and persistence of the population of this endangered subspecies of red squirrel (Rushton et al. 2006). Abert's squirrels are known to reduce cone crops and pine tree growth by clipping branch tips for food (Allred et al. 1994, Allred and Gaud 1994).

**Eastern Fox Squirrel (*Sciurus niger*)**

The eastern fox squirrel (*Sciurus niger*) is native to the eastern and central US and southern Canada (Hall 1981, Koprowski 1994b), and has been introduced in the western US and Canada (Long 2003). *S. niger* is now found in numerous locations throughout Idaho, occurring in urban and agricultural areas, and have expanded their range since initial introduction in the state and neighboring Washington (Marshall 1941, Wright and Weber 1979, Larrison and Johnson 1981, Koprowski 1994b). The squirrel is also found in Asotin County and the Seattle area of Washington (Flahaut 1941, Yocom 1950). This species is currently found in eight counties in Oregon, but is primarily restricted to urban areas and nut tree orchards (Flyger and Gates 1982, Verts and Carraway 1998). *S. niger* can now be found in many urban areas and woodlands in the Central Valley and Coastal Range of California due to numerous introductions (Jameson and Peeters 1988). Although the species occurs naturally in Colorado, many populations are present because of introductions in the early 1900s (Fitzgerald et al. 1994). Fox squirrels are also expanding their range naturally in Colorado; following riparian corridors, invading urbanized areas, and inhabiting areas newly cultivated with deciduous trees (Fitzgerald et al. 1994, Geluso 2004). Fox squirrels are present in Hondo, Roswell, and Carlsbad, New Mexico and western Texas presumably because of introductions (Frey and Campbell 1997), however the date and source of the introductions is unknown (Findley 1987). The eastern fox squirrel was also introduced to Pelee Island on Lake Erie in Ontario, Canada (Peterson 1966).

Fox squirrels are able to invade riparian areas, oak woodlands, and areas with increased edge, tolerate low tree density (Wolf and Roest 1971, Fitzgerald et al. 1994, Verts and Carraway 1998). Fox squirrels will readily consume a variety of agricultural products including many fruits and nuts (Wolfe and Roest 1971, Flyger and Gates 1982, Koprowski 1994b), and will nest in introduced *Eucalyptus globulus* and the attics of private homes (Wolf and Roest 1971, Koprowski 1994b, Verts and Carraway 1998). The impact of *S. niger* on the native western gray and Douglas squirrels (*Tamiasciurus douglasii*) on the western US coast is attributed to competition for resources, and is a growing conservation concern (Ingles 1947, Robinson and McTaggart-Cowan 1954). Abert's squirrels in the Black Forest of Colorado have experienced a population decline attributed to competition with introduced fox squirrels (Fitzgerald et al. 1994).

**Eurasian Red Squirrel (*Sciurus vulgaris*)**

The Eurasian red squirrel occurs naturally from the Iberian Peninsula and Great Britain east to Kamchatka Peninsula in Russia and Japan, while occurring as far south as the Mediterranean and Black Seas, parts of Mongolia and China (Wilson and Reeder 1993).

*S. vulgaris* has been successfully introduced to a few areas just outside of native range in Europe and Asia, including Russia and many independent republics of the former U.S.S.R., as well as Germany (Long 2003). Re-introduction projects have been implemented to re-establish populations in parts of Great Britain (Long 2003). The squirrel is known to strip the bark from conifers, which can damage plantations (Long 2003).

**Pallas's Squirrel (*Callosciurus erythraeus*)**

Pallas's squirrel is native to Southeast Asia (Wilson and Reeder 1993), and has been introduced to Argentina west of Buenos Aires, France, and Japan (Aprile and Chicco 1999, Gurnell and Wauters 1999, Long 2003). Pallas's squirrel damages trees utilized for agriculture, silviculture, and horticulture in both Argentina and Japan (Guichon et al. 2005). Squirrels have been observed eating native as well as domestic bird eggs, and they destroy irrigation systems used to provide water for tree plantations and poultry farms in Argentina (Setoguchi 1990, Guichon et al. 2005, Tamura and Ohara 2005). Dispersal and subsequent spread of the species appears facilitated by anthropogenic environments (Guichon et al. 2005, Miyamoto et al. 2004), and are also spread in Argentina by people who find them attractive (Guichon et al. 2005). Continued spread in Japan is of conservation concern as the species will likely encounter *Sciurus lis*, a declining tree squirrel native to Japan (Miyamoto et al. 2004). The density of the introduced population in France is now higher than the native *S. vulgaris* (Gurnell and Wauters 1999).

**Finalayson's Squirrel (*Callosciurus finlaysoni*)**

Finalayson's squirrel is native to Southeast Asia including Burma, Cambodia, Laos, and Vietnam (Wilson and Reeder 1993). Two pairs of squirrels were introduced to an urban park in Acqui Terme, Italy in 1981 (Bertolino et al. 1999). The species persists at the park with an estimated population between 40-50 squirrels (Bertolino et al. 2004).

The squirrels have been observed consuming many parts of native plants, and are known to strip bark (Bertolino et al. 1999, Bertolino et al. 2004).

**Northern Palm Squirrel (*Funambulus pennantii*)**

The northern palm squirrel is native to portions of the Middle East and India (Wilson and Reeder 1993), and was introduced to the Australian cities of Perth, Western Australia and Sydney, New South Wales (Long 2003). The population in Sydney has since been extirpated (Long 2003). The squirrel is known to eat citrus and stone fruits, and also chew on electrical wiring in and around the city of Perth (Long 2003).

**American red squirrel (*Tamiasciurus hudsonicus*)**

The American red squirrel is native to boreal forests of Canada and the northern US as well as occupying the Cascade, Rocky, and Appalachian Mountains (Steele 1998). *T. hudsonicus* was introduced to many islands associated with Alaska and Canada including Newfoundland, Vancouver Island, the Queen Charlotte Islands, Kodiak Island, Afognak Islands, and the Alexander Archipelago (Payne 1976, Long 2003). Negative impacts from this species have not been reported, and the populations have limited ability to disperse as a result of the insular introductions (Long 2003).

**Mexican Red-bellied Squirrel (*Sciurus aureogaster*)**

The Mexican red-bellied squirrel is native to Mexico and northern Guatemala (Wilson and Reeder 2003), and was introduced to Elliott Key, Florida, in 1938 by a resident of that island (Brown and McGuire 1969, Layne 1997, Long 2003). Squirrels became common on the island, and were reported on nearby Sands Key and Adams Key, likely a result of natural dispersal from individuals swimming (Brown and McGuire 1969, Layne 1997). One individual was captured while swimming across Caesar's Creek in the direction of Old Rhodes Key to the south of Elliott Key (Layne 1997). The population was considered extirpated by Hurricane Andrew in 1992 (Layne 1997), however Koprowski et al. (2005) confirmed the squirrel's persistence on Elliott Key.

Mexican red-bellied squirrels relied heavily upon introduced plant species for food sources including the coconut palm (*Cocos nucifera*), Australian pine (*Casuarina equisetifolia*), and sapodilla (*Manilkara zapota*) as well as consuming papaya (*Carica papaya*) to a lesser extent (Brown and McGuire 1973, Brown and McGuire 1975). However, beginning in the early 1990s, the National Park Service implemented a plan to eradicate non-native plants from all of the islands in Biscayne National Park. The removal of these plants eliminated major food sources for the squirrel, forcing *S. aureogaster* to rely exclusively on native plants. We documented the squirrels consuming numerous native seeds including mahogany pods (*Swietenia mahagoni*), tamarind pods (*Lysiloma latisiliqua*), blackbead (*Pithecellobium unguis-cati*), and Florida poisonwood berries (*Metopium toxiferum*). The squirrels also clipped numerous branches when seeds were unavailable and consumed the phloem of the plant, especially from wild mastic trees (*Masitchodendron foetidissimum*). Tilmant (1980) found squirrels utilizing thatch palm leaves (*Thrinax* spp.) in lining their nests, while Brown and McGuire (1975) noted that the majority of nest material came from the nest tree itself. All of these factors demonstrate a negative impact on native plants from the exotic squirrel on Elliott Key.

We were also concerned with the impact of Mexican red-bellied squirrels on native fauna. The white-crowned pigeon (*Columba leucocephala*) occurs in Biscayne National Park, is listed as threatened by the Florida Fish and Wildlife Conservation Commission (FFWCC). *C. leucocephala* relies on many of the same food resources as the introduced squirrel such as Florida poisonwood and pigeon plum (*Coccoloba diversifolia*) berries (Bancroft and Bowman 1994, FFWCC 2004). Also occurring within Biscayne National Park is the Florida tree snail (*Liguus fasciatus*), a state species of special concern (Florida Fish and Wildlife Conservation Commission 2004). This snail may serve as a food resource for *S. aureogaster* occasionally. Although this population of *S. aureogaster* was not observed raiding bird nests, other *Sciurus* species take bird's eggs and nestlings as food items (Koprowski 1994a). The squirrel's ability to disperse to other islands is also of great concern as *S. aureogaster* will encounter more opportunities to impact native floral and fauna as well as agriculture. Two native tree squirrels occur in Florida, and dispersing Mexican red-bellied squirrels would be an unwelcome source of competition for these species. *S. aureogaster* would encounter *S. carolinensis* upon reaching Key Largo (~8.5 km straight line distance from Elliott Key) or the mainland (~12.5 km straight line distance from Elliot Key, Brown 1997). The Big Cypress fox squirrel (*Sciurus niger*

*avicennia*) is the nearest population of eastern fox squirrels to Elliott Key at approximately 84 km (Hall 1981, Brown 1997). This is of particular concern as this subspecies is listed as threatened by the State of Florida, and resource needs of the threatened fox squirrel and the red-bellied squirrel would overlap greatly (Brown and McGuire 1975, Brown 1997, Florida Fish and Wildlife Conservation Commission 2004).

The National Park Service has implemented a plan to eradicate this species from Biscayne National Park as the species poses a threat to native species and agriculture. Squirrel nests will be marked with reflective forestry flagging, and trained personnel will return to each nest at night when squirrels are likely occupying the nest, and discharge non-toxic shot from a shotgun into the nest. Nesting material will be completely removed from the nest tree, and the squirrel carcasses collected for further study. Nest boxes will be erected as both a monitoring tool and an additional way to capture individual squirrels that came to reside in the nest boxes.

## REASONS FOR SUCCESS

Tree squirrels are able to establish viable populations from a very small number of propagules (Brown and McGuire 1975, Aprile and Chicco 1999, Long 2003). Success rates of introductions appear to be extraordinarily high, even from small (< 20 individuals) starting populations (Wood et al. 2007). Most tree squirrels reproduce twice a year with a mean litter size between 2 and 4 individuals (Heaney 1984, Koprowski 1994a, Koprowski 1994b), so populations can experience rapid growth after initial introduction to novel environments. Tree squirrels are long lived, with individuals living between 5 and 15 years (Koprowski 1994a, Koprowski 1994b). The ability to disperse is important to the spread of an invasive species, and tree squirrels are capable of dispersing great distances, with squirrels known to cross waterways (Layne 1997), agricultural and urban areas (Jameson and Peeters 1988). Tree squirrels are capable of using a wide variety of plant and animal material for food, and possess strong jaw muscles enable them to open most seeds and nuts, which are excellent sources of fats and proteins (Steele and Koprowski 2001). Nests are constructed for protection against the elements, and can either be a stick and leaf nest (drey) or cavity in a tree (Koprowski 1994a, Koprowski 1994b). Many species of tree squirrel have the ability to adapt to human-impacted environments (Jameson and Peeters 1988, Geluso 2004,).

## MANAGEMENT

In some cases, introduced populations of tree squirrels pose such a great threat to conservation and economic interests to warrant control or eradication. In Great Britain, *S. carolinensis* has been subject to a wide range of control and eradication tactics including trapping, shooting, and poisoning (Dagnall et al. 1998). A grass-roots movement by farmers and woodland owners initiated the first concerted effort to reduce squirrel numbers, with hundreds of "Grey Squirrel Hunting Clubs" springing up to exterminate an estimated 100,000 squirrels by 1947 (Sheail 1999). The British government offered free cartridges to participating clubs in support of the eradication effort, and promoted an "anti-grey squirrel" campaign over BBC radio (Sheail 1999). In 1952, the government decided to offer bounties on squirrel tails, a program that lasted until 1957, and resulted in 1,000,000 squirrels being turned in (Sheail 1999). The bounty program was ended because the squirrel population had not been reduced to a manageable level, with some suggesting that the problem had even grown worse despite the effort (Sheail 1999), however, whether the removal of so many individuals slowed their population expansion is not known. Government support returned in the form of the Squirrel (Warfarin) Order of 1973, which initiated a poisoning program that was deemed most effective for squirrel control (Dagnall et al. 1998, Sheail 1999). Increasing pressure from animal right's groups against poisoning of squirrels despite the squirrel's invasive nature has given rise to new ideas on controlling *S. carolinensis* in Great Britain (Dagnall et al. 1998). Manipulation of the physical environment, namely managing forests in a way that will reduce squirrel numbers, and current research into sterilization techniques that will involve a vaccine induced immuno-contraceptive for squirrels have been recent focuses of research for grappling with the *S. carolinensis* problem (Dagnall et al 1998). Eastern gray squirrels were the focus of an eradication attempt in Italy that was initially successful but halted because of objection from animal rights groups (Bertolino and Genovesi 2003).

## MONITORING

The continual monitoring of introduced squirrel populations will be important in understanding impacts that invasives have in their new location and to detect range expansions. Numerous monitoring techniques facilitate these goals. Nest boxes attract squirrels seeking temporary refuge and also to build nests for long term use (Brown and McGuire 1975), and are an important method to determining success of an eradication effort or to monitor range expansion as squirrels colonize new areas. Hair tubes or snares collect hairs from individual squirrels which is helpful to determine presence or absence in areas that are in danger of being colonized by a spreading population (Gurnell et al. 2004a). Transect surveys through forested areas in or near sites of introduction can determine squirrel density and distribution (Brown and McGuire 1975, Koprowski et al. 2005, Koprowski et al. 2006).

## CONCLUSION

Understanding the impacts created by invasive tree squirrel populations is important for the conservation and continuation of native species, especially native tree squirrels. As discussed herein, invasive tree squirrels can affect the environments in a multitude of ways, and documentation of these impacts is important to enable justification of control and eradication programs. Assessment of current distributions and knowledge of an invasive population's ecology is crucial in predicting the path of spread and planning ways to possibly slow or eliminate population expansion. The continued growth in global interest in invasive species and the rapid anthropogenic spread of tree squirrels make these species excellent model systems for the study of impacts. Furthermore, the nearly cosmopolitan distribution, questionable conservation status, and poor knowledge of native tree squirrel species (Koprowski and Rajamani In Press) make studies of the tree squirrels imperative.

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## LITERATURE CITED

ALLRED, W. S., AND W. S. GAUD. 1994. Effects of Abert squirrel herbivory on foliage and nitrogen losses in ponderosa pine. Southwestern Naturalist 39:350-353.

ALLRED, W. S., W. S. GAUD, AND J. S. STATES. 1994. Effects of herbivory by Abert'ssquirrels (*Sciurus aberti*) on cone crops of ponderosa pine. Journal of Mammalogy 75:700-703.

APRILE, G., AND D. CHICCO. 1999. Nueva especie exotica de mamifero en la Argentina: la ardilla la de vientre rojo (*Callosciurus erythraeus*). Mastozoologia Neotropical 6:7-14.

BANCROFT, G. T., AND R. BOWMAN. 1994. Temporal patterns in diet of nestling white-crowned pigeons: implications for conservation of frugivorous columbids. Auk 111:844-852.

BANFIELD, A. W. F. 1974. The mammals of Canada. University of Toronto Press, Toronto, Canada.

BAYRAKCI, R., A. B. CAREY, AND T. M. WILSON. 2001. Current status of the western gray squirrel population in the Puget Trough, Washington. Northwest Science 75:333-341.

BERTOLINO, S., I. CURRADO, AND P. J. MAZZOGLIO. 1999. Finlayson's (variable) squirrel *Callosciurus finlaysoni* in Italy. Mammalia 63:522-525.

BERTOLINO, S., AND P. GENOVESI. 2003. Spread and attempted eradication of the grey squirrel (*Sciurus carolinensis*) in Italy, and consequences for the red squirrel (*Sciurus vulgaris*) in Eurasia. Biological Conservation 109:351-358.

BERTOLINO, S., P. J. MAZZOGLIO, M. VAIANA, AND I. CURRADO. 2004. Activity budget and foraging behavior of introduced *Callosciurus finlaysoni* (Rodentia, Sciuridae) in Italy. Journal of Mammalogy 85:254-259.

BERTOLINO, S., AND P. GENOVESI. 2005. The application of the European strategy on invasive alien species: an example with introduced squirrels. Hystrix: Italian Journal of Mammalogy 16:59-69.

BROWN, L.N., AND R.J. MCGUIRE. 1969. Status of the red-bellied squirrel (*Sciurus aureogaster*) in the Florida Keys. American Midland Naturalist 82:629-630.

BROWN, L. N. 1997. Mammals of Florida. Windward Publishing Company, Minneapolis, USA.

BROWN, L. N. AND R. J. MCGUIRE. 1973. Coconut feeding behavior of the red-bellied squirrel on Elliot Key, Dade County, Florida. American Midland Naturalist 89:498.

Brown, L. N., and R. J. McGuire. 1975. Field ecology of the exotic Mexican red-bellied squirrel in Florida. Journal of Mammalogy 56:405-419.

Bryce, J. M., J. R. Speakman, P. J. Johnson, and D. W. Macdonald. 2001. Competition between Eurasian red and Eastern grey squirrels: the energetic differences of body mass. Proceedings of the Royal Society of London 268:1731-1736.

Dagnall, J., J. Gurnell, and H. Pepper. 1998. Bark-stripping damage by gray squirrels in state forests of the United Kingdom: a review. Pages 249-261 *in* M. A. Steele, J. F. Merritt, and D. A. Zegers, editors. Ecology and evolutionary biology of tree squirrels. Special Publication, Virginia Museum of Natural History, Martinsville, USA.

Davis, R., and D. E. Brown. 1988. Documentation of the transplanting of Abert's squirrels. Southwestern Naturalist 32:490-492.

Edelman, A. J., and J. L. Koprowski. 2005. Diet and tree use of Abert's squirrels (*Sciurus aberti*) in a mixed-conifer forest. The Southwestern Naturalist 50:461-465.

Edelman, A. J., and J. L. Koprowski. 2006. Characteristics of Abert's squirrel (*Sciurus aberti*) cavity nests. The Southwestern Naturalist 51:64-70.

Findley, J. S. 1987. The natural history of New Mexican mammals. University of New Mexico, Albuquerque, USA.

Fitzgerald, J. P., C. A. Meaney, and D. M. Armstrong. 1994. Mammals of Colorado. University Press of Colorado, Boulder, USA.

Flahaut, M. R. 1941. Exotic squirrels in the Seattle area. The Murrelet 22:63-64.

Florida Fish and Wildlife Conservation Commission. 2004. Florida's endangered species, threatened species, and species of special concern. Florida Fish and Wildlife Conservation Commission, Tallahassee, USA.

Flyger, V., and J. E. Gates. 1982. Fox and gray squirrels. Pages 209-229 *in* J. A. Chapman and G. A. Feldhamer, editors. Wild mammals of North America. Johns Hopkins University, Baltimore, USA.

Frey, J. K., and M. L. Campbell. 1997. Introduced populations of fox squirrel (*Sciurus niger*) in the Trans-Pecos and Llano Estacado regions of New Mexico and Texas. Southwestern Naturalist 42:356-358.

Geluso, K. 2004. Westward expansion of the eastern fox squirrel (*Sciurus niger*) in northeastern New Mexico and southeastern Colorado. Southwestern Naturalist 49:111-116.

Guichon, M. L., M. Bello, and L. Fasola. 2005. Expansion poblacional de una especie introducida en la Argentina: la ardilla de vientre rojo *Callosciurus erythraeus*. Mastozoologia Neotropical 12:189-197.

Gurnell, J., and L. Wauters. 1999. *Callosciurus erythraeus* (Pallas, 1779). Pages 182-183 *in* A. J. Mitchell-Jones, editor. The atlas of European mammals. Academic Press, London, United Kingdom.

Gurnell, J., P. W. W. Lurz, M. D. F. Shirley, S. Cartmel, P. J. Garson, L. Magris, and J. Steele. 2004a. Monitoring red squirrels *Sciurus vulgaris* and grey squirrels *Sciurus carolinensis* in Britain. Mammal Review 34:51-74.

Gurnell, J., L. A. Wauters, P. W. W. Lurz, and G. Tosi. 2004b. Alien species and interspecific competition: effects of eastern grey squirrels on red squirrel population dynamics. Journal of Animal Ecology 73:26-35.

Haagner, A. 1920. South African mammals. H. F. & G. Witherby, London, United Kingdom.

Hall, E. R.. 1981. The mammals of North America. Volume 2. The Ronald Press Company, New York, USA.

Heaney, L. R. 1984. Climatic influences on life-history tactics and behavior of North American tree squirrels. Pages 43-78 *in* J. O. Murie and G. R. Michener, editors. The biology of ground-dwelling squirrels: annual cycles, behavioral ecology, and sociality. University of Nebraska, Lincoln, USA.

Hibbard, A. H. 1956. Range and spread of the gray and fox squirrels in North Dakota. Journal of Mammalogy 37:525-531.

Hoffman, R. S., P. L. Wright, and F. E. Newby. 1969. The distribution of some mammals in Montana. I. Mammals other than bats. Journal of Mammalogy 50:579-604.

Huey, L. M. 1964. The mammals of Baja California, Mexico. Transactions of the San Diego Society of Natural History 13:85-168.

Hutton, K. A., J. L. Koprowski, V. L. Greer, M. I. Alanen, C. A. Schauffert, and P. J. Young. 2003. Use of mixed-conifer and spruce-fir forests by an introduced population of Abert's squirrels. The Southwestern Naturalist 48:257-260.

Ingles, L. G. 1947. Ecology and life history of the California gray squirrel. California Fish and Game 33:139-158

Jameson Jr., E. W., and H. J. Peeters. 1988. Mammals of California. University of California, Berkeley, USA.

Koprowski, J. L. 1994a. *Sciurus carolinensis*. Mammalian Species 480:1-9.

Koprowski, J. L. 1994b. *Sciurus niger*. Mammalian Species 479:1-9.

Koprowski, J. L., G. T. Kellison, and S. L. Moneysmith. 2005. Status of red-bellied squirrels (*Sciurus aureogaster*) introduced to Elliott Key, Florida. Florida Field Naturalist 33:128-129.

Koprowski, J. L., N. Ramos, B. S. Pasch, and C. A. Zugmeyer. 2006. Observations on the ecology of the endemic Mearns's squirrel (Tamiasciurus mearnsi). The Southwestern Naturalist 51:426-430.

KOPROWSKI, J. L., AND N. RAJAMANI. In Press. Global hotspots, centers of diversity, and conservation of the tree and flying squirrels. Current Science 92.

KRAMER, R. J. 1971. Hawaiian land mammals. Charles E. Tuttle Co., Rutland, USA.

LARRISON, E. J., AND D. R. JOHNSON. 1981. Mammals of Idaho. The University Press of Idaho, Moscow, USA.

LAYNE, J. 1997. Nonindigenous mammals. Pages 157-186 *in* D. Simberloff, D. Schmitz, and T. Brown, editors. Strangers in paradise: impact and management of nonindigenous species in Florida. Island Press, Washington, D.C., USA.

LEVER, C. 1985. Naturalized mammals of the world. Longman, London, United Kingdom.

LONG, J. L. 2003. Introduced mammals of the world. CABI Publishing, Wallingford, United Kingdom.

MACDONALD, I. A. W., D. M. GRABER, S. DEBENEDETTI, R. H. GROVES, AND E. R. FUENTES. 1988. Introduced species in nature reserves in Mediterranean-type climatic regions of the world. Biological Conservation 44:37-66.

MARSHALL, W. H. 1941. The fox squirrel in Idaho. The Journal of Mammalogy 22:86-87.

MCINNES, C. J., A. R. WOOD, K. THOMAS, A. W. SAINSBURY, J. GURNELL, F. J. DEIN, AND P. F. NETTLETON. 2006. Genomic characterization of a novel poxvirus contributing to the decline of the red squirrel (*Sciurus vulgaris*) in the UK. Journal of General Virology 87:2115-2125.

MIYAMOTO, A., N. TAMURA, K. SUGIMURA, AND F. YAMADA. 2004. Predicting habitat distribution of the alien formosan squirrel using logistic regression model. Global Environmental Research 8:13-21.

MOUNTFORD, E. P. 1997. A decade of grey squirrel bark stripping damage to beech in Lady Park Wood, UK. Forestry 70:17-29.

MOUNTFORD, E. P. 2006. Long-term patterns and impacts of grey squirrel debarking in Lady Park Wood young-growth stands (UK). Forest Ecology and Management 232:100-113.

PAYNE, N. F. 1976. Red squirrel introduction to Newfoundland. The Canadian Field-Naturalist 90:60-64.

PETERSON, R. L. 1966. The mammals of eastern Canada. Oxford University Press, Toronto, Canada.

RAYDEN, T. J., AND P. S. SAVILL. 2004. Damage to beech woodlands in the Chilterns by the grey squirrel. Forestry 77:249-253.

ROBINSON, D. J., AND I. MCTAGGART-COWAN. 1954. An introduced population of the gray squirrel (*Sciurus carolinensis* Gmelin) in British Columbia. Canadian Journal of Zoology 32:261-282.

RUSHTON, S. P., P. W. W. LURZ, J. GURNELL, AND R. FULLER. 2000. Modelling the spatial dynamics of parapoxvirus disease in red and grey squirrels: a possible cause of the decline in the red squirrel in the UK? Journal of Applied Ecology 37:997-1012.

RUSHTON, S. P., D. J. A. WOOD, P. W. W. LURZ, AND J. L. KOPROWSKI. 2006. Modelling the population dynamics of the Mt. Graham red squirrel: can we predict its future in a changing environment with multiple threats? Biological Conservation 131:121-131.

RYAN, L. A., AND A. B. CAREY. 1995. Distribution and habitat of the western gray squirrel (*Sciurus griseus*) on Ft. Lewis, Washington. Northwest Science 69:204-216.

SEEBECK, J. H. 1995. Eastern grey squirrel. Page 234 *in* P. W. Menkhorst, editor. Mammals of Victoria. Oxford University Press, Melbourne, Australia.

SETOGUCHI, M. 1990. Food habits of red-bellied tree squirrels on a small island in Japan. Journal of Mammalogy 71:570-578.

SHEAIL, J. 1999. The grey squirrel (*Sciurus carolinensis*)-a UK historical perspective on a vertebrate pest species. Journal of Environmental Management 55:145-156.

SCHMIDLY, D. J. 2004. The mammals of Texas. University of Texas, Austin, Texas, USA.

SIGNORILE, A. L., AND J. EVANS. 2007. Damage caused by the American grey squirrel (*Sciurus carolinensis*) to agricultural crops, poplar plantations and semi-natural woodland in Piedmont, Italy. Forestry 80:89-98.

STEELE, M. A. 1998. *Tamiasciurus hudsonicus*. Mammalian Species 586:1-9.

STEELE, M. A., AND J. L. KOPROWSKI. 2001. North American tree squirrels. Smithsonian Institution, Washington D.C., USA.

TAMURA, N., AND S. OHARA. 2005. Chemical components of hardwood barks stripped by the alien squirrel Callosciurus erythraeus in Japan. Journal of Forestry Research 10:429-433.

TILMANT, J. T. 1980. Investigations of rodent damage to the thatch palms *Thrinax morrisii* and *Thrinax radiata* on Elliot Key, Biscayne National Park, Florida. South Florida Research Center, Everglades National Park, USA.

UNITED STATES FISH AND WILDLIFE SERVICE. 1993. Mount Graham red squirrel recovery plan. United Sates Fish and Wildlife Service, Albuquerque, USA.

VERTS, B. J. AND L. N. CARRAWAY. 1998. Land mammals of Oregon. University of California, Berkley, USA.

WASHINGTON DEPARTMENT OF WILDLIFE. 1993. Status of the western gray squirrel (*Sciurus griseus*) in Washington. Final status report. Olympia, Washington, USA.

WAUTERS, L. A., P. W. W. LURZ, AND J. GURNELL. 2000. The effects of interspecific competition by grey squirrels (*Sciurus carolinensis*) in the space use and population dynamics of red squirrels (*S. vulgaris*) in conifer plantations. Ecological Research 15:271-284.

Wilson, D. E., and D. M. Reeder. 1993. Mammal species of the world. Smithsonian Institution, New York, USA.

Wolfe, T. F., and A. I. Roest. 1971. The fox squirrel (*Sciurus niger*) in Ventura County. California Fish and Game 57:219-220.

Wood, D. J. A., J. L. Koprowski, P. W. W. Lurz. 2007. Tree squirrel introduction: a theoretical approach with population viability analysis. Journal of Mammalogy 88:1271-1279.

Wright, G. M. and J. W. Weber. 1979. Range extensions of the fox squirrel in southeastern Washington and into adjacent Idaho. The Murrelet 60:73-75.

Yocom, C. F. 1950. Fox squirrels in Asotin County, Washington. The Murrelet 31:34.

# INVASIVE RATS AND BUBONIC PLAGUE IN NORTHWEST UGANDA

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***Abstract:*** Major introductions of roof rats (Rattus rattus) likely occurred in the ports of East Africa during the Third Plague Pandemic in the late 1800's. Transport via trains, boats, and trucking likely introduced this species to inland areas of East Africa, ultimately including Northwest Uganda. Historic plague outbreaks occurred during the early part of the 20th century and continue to cause a human disease burden in the West Nile region of NW Uganda via the bubonic, septicemic, and pneumonic forms. Four field sites in this area were trapped to determine the rodent species composition in commensal and peridomestic areas of villages and associated flea burdens of the rodents. Rattus rattus were the most prevalent rodent trapped in commensal areas followed by the Nile rat (Arvicanthis niloticus). The most common peridomestic species of rodent was the Nile rat. Other peridomestic rodent species captured included, Mastomys natalensis, Lophuromys flavopunctatus, L. sikapusi, Gerbil (Tatera spp.), Lemniscomys spp., and 4 unknown species. Flea burdens on commensal R. rattus averaged 1.7±1.2 fleas per animal and on all peridomestic rodents, average 2.0±0.7 per animal. Additionally, commensal areas were sampled to determine free-living flea populations. Burrow swabbing indicated an average 0.19±0.12 fleas/burrow. Lighted flea traps averaged 1.3±0.6 fleas per household and dark flea traps averaged 0.5±0.4 fleas per household.

***Key Words:*** flea vector, invasive species, plague, *Rattus rattus*, rodents, Uganda, *Yersinia pestis*.

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## INTRODUCTION

As described by researchers in the Democratic Republic of the Congo (DRC), "Plague is a disease of rats, a variegated gathering of rodents with different degrees of tolerance and sensitiveness to *Yersinia pestis*, living in frail equilibrium" (translation from Dutch, Janssens and Pattyn 1994). More than any of the 1,814 (order Rodentia, Nowak 1991) known rodents in the world, *Rattus norvegicus*, and especially *Rattus rattus*, have repeatedly been the tipping point in this "frail equilibrium" of the enzootic cycle of bubonic plague. The recent historic and current yearly incidence of plague in Northwest Uganda is most likely influenced by the introduction of the non-native *R. rattus* to this area.

Although a simplified explanation, three biovars of plague (*Y. pestis*) caused the three great plague pandemics. The Justinian plague of the sixth century was caused by biovar, *antiqua*, the 16th century Black Death by *medievalis* and the Third/Modern pandemic by the *orientalis* biovar. The Justinian plague occurred between 542-546 in Asia, Africa and Europe and affected 100 million people (Zietz and Dunkelberg 2004). This pandemic likely originated in East Africa, traveled north along the Nile or via oceanic shipping routes from East Africa to Egypt, then Palestine and throughout the Mediterranean region. This pandemic was likely spread by flea-borne transmission of the disease and was the source of epidemics in the following two centuries (Politzer 1954, Gage 1998, WHO 2000, Zietz and Dunkelberg 2004). Keyes (1999) theorizes that a great climatic upheaval occurred during this period causing unusually cool and wet conditions in the Mediterranean region. These climatic events are thought to have allowed environmental conditions that favored the transfer and expansion of fleas among many rodent hosts and led to the spread of the disease.

The second pandemic, known as the "Black Death," occurred in the fourteenth century and caused 50 million deaths, half in Europe and half in Africa and Asia. This plague pandemic likely originated in Central Asia among marmots and

traveled along the Silk Road to Europe (McEvedy 1988). Unlike the first pandemic, this pandemic was driven not only by flea-borne transmission of the disease, but also by pneumonic transfer of the disease among humans. This pandemic was also the source of subsequent smaller plague outbreaks in Europe and Africa in the following centuries, often with equal ferocity (Politzer 1954, Gage 1998). The last large pandemic, called the "Modern Pandemic" began in Hong Kong in 1894 and spread rapidly throughout the world by rats carried on merchant steamships. Within 10 years, plague had entered 77 ports on six continents: 31 in Asia, 12 in Europe, 8 in Africa, 4 in North America, 15 in South America and 7 in Australia. Epidemics resulted on every affected continent and caused over 12 million deaths, most in India. Scientific study during this last pandemic resulted in the identification of the causative agent, *Y. pestis*, by Alexander Yersin in 1894 in Hong Kong and also identified fleas as the primary vector of plague, as well as the role of commensal rats in the outbreaks of human plague (Politzer 1954, Gage 1998, Tikhomirov 1999).

In the last few decades, Africa reported the highest percentage of plague cases in the world and the largest number of human mortalities due to the disease (WHO 2004). During the period of 1980-1997, 19,349 cases (1781 deaths) were reported in Africa–representing 66.8% (75.8%) of the world's totals (WHO 2000). Most outbreaks, during this period, occurred in Eastern, Central and Southern Africa in the countries of Tanzania, South Africa, Madagascar, DRC, Zimbabwe, Mozambique, Malawi, Namibia, Kenya and Uganda (WHO 2000). Cases are rarely reported in northern Africa, but the disease recently reemerged in Algeria, 50 years after its last occurrence with the discovery of previously unknown natural foci (WHO 2004). On the island of Madagascar, during the 18-year period from 1980-1997, 5,986 cases with 493 deaths were recorded (Gratz 1999, Tikhomirov 1999).

## ARRIVAL OF PLAGUE TO WEST NILE REGION OF UGANDA

The plague foci in East/Central Africa are very ancient and were definitely present before British Occupation in the late 1800s (Barrett 1933). Although a small number of researchers theorized that plague evolved in Central Africa (Panagiotakopulu 2004), most authors think *Y. pestis* evolved from the Central Asiatic plateau (Guiyoule et al. 1994). Genetic analysis of biovars and ribotypes of *Y. pestis* agree with a Central Asiatic origin, as Asia has greater diversity of biotypes (Guiyoule et al. 1994). It is thought that biovar, *antiqua*, was introduced to Central Africa from Central Asia at some point before the sixth century, as there were reports of the introduction of plague into North Africa during the first century (Keyes 1999). Many rodent and their flea species throughout Central Asia, the Middle East and North Africa can harbor and transmit plague (Shenbrot and Krasnov 1999). Known as the Great Palaeartic Desert Belt, this desert area extends from the Atlantic coast of northwest Africa in the west to northeastern China. This is an area of high rodent and flea species richness (Shenbrot and Krasnov 1999). It is possible that these rodent complexes, including gerbils and jirds, may have transferred *Y. pestis* from Central Asia to North Africa and subsequently to Central Africa. Gerbils (represented across the entire belt) and jirds harbor numerous complexes of flea species and host switching of fleas among these rodents is common (Shenbrot and Krasnov 1999, Krasnov and Shenbrot 2002). Other authors think the disease may have been introduced from Egypt, Arabia and India by various medieval traders including slave and ivory caravans and/or via pilgrims to and from Mecca (Kilonzo 1976). These caravan routes extended from the Buganda Kingdom in Uganda where plague was or became endemic (Msangi 1975). Additionally, an epidemic of plague was known to have occurred in 1697 in Mombasa, Kenya, which held back a Portuguese attack against the Arabs for 33 months (Twigg 1978).

*Y. pestis*, biovar *orientalis*, was likely introduced into East Africa during the third plague pandemic, but currently appears to be restricted to East African ports (Guiyole et al. 1994). In the DRC, only biovar *antiqua* has been reported and this ancient biovar continues to be most prevalent inland in east/central Africa (Guiyole et al. 1994).

The earliest recorded accounts of plague in Uganda were by missionaries in 1877 (Orochi-Orach 2002). At the time, many indigenous ethnic groups were familiar with the disease and, in their traditional languages, had words for plague (Davis et al. 1968). The Buganda in Uganda called the disease "kampuli," in the Lendu language "zukpa" and in Swahili was called "tauni" (from the Arabic tā'ūn meaning the bubonic form of plague; Msangi 1975, Conrad 1982, Orochi-Orach 2002). In Uganda, between 1910-1946, over 61,000 people contracted the disease in the central districts of Buganda, Busoga, Mbale, Teso and Lango.

Records from the rest of the country were scarce (Hopkins 1949). Mortalities caused by plague during this period were high because treatment access in rural areas where the epidemics were confined was poor. The rodents responsible for the plague outbreaks in these areas were primarily *R. rattus*, *Mastomys natalensis* and *Arvicanthis niloticus*. The major rodent flea vectors responsible for the *Y. pestis* transmission was *X. bransilensis* and *X. cheopis* (Hopkins 1949, Politzer 1954).

From 1889-1954, the main plague foci in Uganda were localized in following areas: the Ituri forest, around Lake Albert, Burungu Island in Lake Victoria, Rubaga hill near Kampala and the Kyaggwe areas in the Districts of Mukono, Masaka, Rakai, Pallisa, Tororo, and Busia. These districts eventually became quiescent except for 2 villages in the Toro District (Ogen-Odoi 1999, Davis et al. 1968). Currently, the active plague foci are in the West Nile region of northwest Uganda. Plague outbreaks have occurred in Okoro County of the Nebbi District and Vurra County of the Arua District and cases of plague are often reported at the Nyapea Mission Hospital in the Nebbi District (Ogen-Odoi, personal communication). Major reported epidemics of human plague have been recorded numerous times during the last 20 years. Most of the cases have occurred in the Nebbi District of Northwestern Uganda where the disease has been endemic for over 40 years. Outbreaks occurred in 1982 (153 reported cases, 3 deaths), 1986 (340 reported cases, 27 deaths), 1993 (167 reported cases, 18 deaths), 2000 (202 reported cases, 50 deaths) and 2001 (319 reported cases, 42 deaths; Kilonzo 1999, WHO 2003). Although not fully understood in the West Nile region, the incidence of human plague cases in other areas has been observed to increase due to deforestation, the result of military operations, civil unrest, deterioration of sanitary conditions and the displacement of populations (Akiev 1982, WHO 2004). In other areas of Africa, times of civil unrest can cause a breakdown in most health services and reporting (Kilonzo 1999).

The West Nile region in northwest Uganda shares its plague focus with the Democratic Republic of the Congo (DRC) to the east. In the DRC, plague mainly exists in two foci situated in the extreme northeast of the country near the border of Uganda. Named the Lake Albert and Lake Edward foci, the Lake Albert foci likely formed an extension into the West Nile area of Uganda. Cases of plague in the Lake Albert region were first officially recognized in 1928 and prior to the 1950s *Mastomys coucha* [*natalensis*] was the primary rodent found in human habitations (98%). Additionally, *Arvicanthis abyssinicus* [*niloticus*] was also present. There were no early reports of *R. rattus* in this area (Politzer 1954). Unlike the Lake Albert focus, rodents collected in the Lake Edward focus were *R. rattus* (52%), *M. coucha* [*natalensis*] (32%) and *A. abyssinicus* [*niloticus*] (14%; Politzer 1954). This indicates that the introduction of *R. rattus* to the region seemed to displace the position of *M. coucha* [*natalensis*] in the huts. Both *X. cheopis* and *X. bransiliensis* were considered the early vectors of plague in the Lake Albert region and primarily *X. bransiliensis* in the Lake Edward focus (Politzer 1954).

In recent times, most of the plague cases in the West Nile Region were reported from the southern Nebbi District. Beginning in 1998, plague cases began to be reported from the northern Arua District (Orochi-Orach 2002). A US Centers for Disease Control and Prevention (CDC) analysis of local health records indicated that there were 1,610 cases of plague form 1989-2003 in Northwestern Uganda but less than 1% of cases were laboratory confirmed. Plague spread from 23 local parishes to 30 and averaged 298 cases/year during this period. The overall mortality rate was 25.9% with higher mortality from the pneumonic (52.8%) and septicemic (40.5%) forms compared to the bubonic (18.3%). Only 1/3 of cases were admitted into a health facility. Eighty-seven percent of the cases were bubonic, 6.2% pneumonic and 5.3% septicemic. There was female predominance (52.7%) with an average age of 19.0 years (Staples et al. 2004).

Plague cases in the West Nile region are known to be highly underreported (R. Enscore, CDC DVBID, personal communication). Likewise, the WHO concedes that officially reported data on plague do not adequately reflect the true incidence of plague, represents only a portion of the actual number of cases and may not even represent the known, active foci in the world. Global statistics on plague are incomplete because of the reluctance to officially notify plague cases as well as inadequate surveillance and reporting (WHO 2004). In Uganda, cultural and religious beliefs are known to affect the reporting of plague (Orochi-Orach 2002). When a death occurs among the Alur tribe in Northwest Uganda it is often thought to be a result of a bad omen or due to witchcraft. This belief is augmented if inexplicable deaths occur repeatedly as in the case of an epidemic of plague. When rats are observed to have died in a village or

a home, they are often believed to have been "dropped" by a witch-doctor in the village. Often, the belief is that the deaths occur from an "ill spirit" "sweeping the village" in reprisal of an unpaid debt or other reasons (Orochi-Orach 2002).

The peak months of plague in the West Nile region are September through December, corresponding to the rainy season. Although the Nebbi district in the West Nile has been surveyed for the last 13 years, the epizoology of plague continues to be poorly understood (Laudisoit et al. 2007). A possible enzootic cycle in the West Nile region of Uganda is described in Figure 1 (based on: Politzer 1954, Gage 1998, Gratz 1999).

Weather, food presence (including periods of harvest), natural rodent mortalities (epizootic) and excessive human fleabites influence this enzootic cycle. Throughout Africa, excessive rodent population outbreaks often occur after periods of excessive rainfall, especially after periods of drought (Fiedler 1994, Leirs et al. 1996), which promotes excessive growth of vegetation, decreased competition, increased reproduction rates, and increased cover. It is believed that epidemics in some areas of Africa follow an unusually prolonged drought in the area, which may force many field rodents to seek food within human dwellings, thus passing infection to peridomestic and commensal

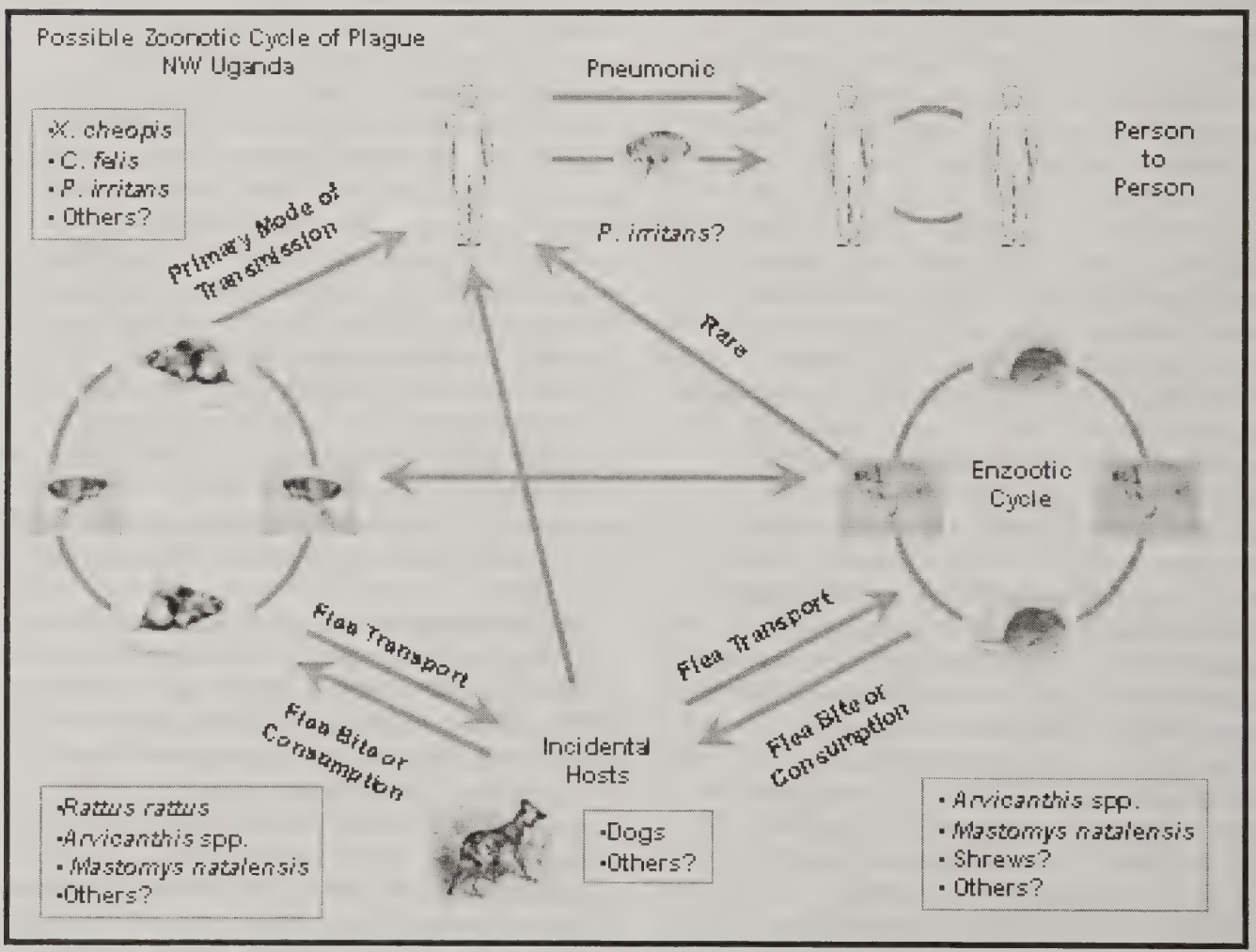



**Figure 1.** Proposed enzootic cycle of plague in the West Nile region of Uganda.

rodents, domestic animals and humans. Observations in this region indicate that nearly all-human epidemics are preceded by moderate to massive epizootics in rodents and excessive fleabites to humans (Ogen-Odoi, personal communication). These observations are yet to be studied in detail.

Additionally, the enzootic cycle is affected by cultural activities. The Alur ethnic group in the West Nile region historically stored harvested grain in granaries outside of the structures used for living and sleeping. Due to increased theft of grain and animals, food and often livestock are largely stored in the same structure where people sleep. Likewise, when a person dies in the Alur culture, people gather and sleep in the home of the deceased for 3-4 days. This activity is based on belief that the deceased spirit lingers and, if ignored by surviving relatives and friends, may later negatively affect crop harvest or cause ill to the survivors (Orachi-Orach 2002). These activities may increase the risk of contracting plague.

## ARRIVAL OF *RATTUS RATTUS* TO THE WEST NILE REGION

Species of the genus *Rattus* have likely been responsible for more human suffering than any other group of vertebrates via their role in transmission of disease and destructive impact on food crops (Alpin et al. 2003). The dispersal of *Rattus* spp. from its native homeland is remarkable in that it occurred in the presence of pre-existing, diverse rodent fauna (Alpin et al. 2003). The presence of numerous, established competitors would normally represent a significant impediment to the establishment of an invasive species, but *Rattus* spp. have not only managed to gain a foothold in diverse habitats throughout the world, but are thriving, including areas in Northwest Uganda. *R. norvegicus* is known to displace *R. rattus*. In general, displacements of *R. rattus* by *R. norvegicus* usually occur in ports and urban sites, but are much less common in the rural areas of developing countries where the *R. rattus* is encountered more frequently (Gage 1998). Such is the case in the West Nile region of Uganda. Originating in India or Burma, *R. rattus* tends to have the advantage in warmer climates while *R. norvegicus* appears to have the advantage in cooler climates (Avery 1985). The first appearances of *R. rattus* in Eastern and Southern Africa seem to have predated the Third Plague Pandemic. Iron Age sites in Zambia have recorded the presence of *R. rattus* around 1500 and 1600 AD, the northern Transval (currently South Africa) around 1000 A.D. and Natal (currently South Africa) in the eighth century (Davis and Fagan 1962, Plug et al. 1979, Voight and Von den Driesch 1984).

Although *R. rattus* may have been present in prior times, major introductions of the rodent to East Africa seemed to have occurred in the late 1800s (Fieldler 1988) from the East. It was proposed that the construction of the railway from the Eastern port of Mombasa, Kenya to Kisumu, Kenya and ultimately into Uganda led to an importation of plague (*Y. pestis* biovar *orientalis*) and *Rattus* spp. from docking ships (Twigg 1978). During the period of 1903-1908 most of the ports in Lake Victoria became involved in plague epidemics, which continued to spread inland (Politzer 1954). Prior to the arrival of *R. rattus*, previous outbreaks of plague must have occurred due to other species of rodents maintaining the enzootic and epizootic cycles (Politzer 1954). During this period prior to the publication of Politzer's book (1954), *M. natalensis* was the most common commensal rodent found in areas of Uganda not invaded by the *R. rattus*. In East Africa, *M. natalensis* was displaced by invading *R. rattus* and augmented the frail equilibrium of plague in these areas (Janssens and Pattyn 1994). *R. rattus* was not observed in the West Nile region as late as 1938 (Anonymous 1939).

Alternatively, the invasion of *R. rattus* into the West Nile region may have arrived from the west. Rats were observed in Northeast DRC in 1909 in what is now the city of Kisingani, and one report suggested the presence of this species slightly East of this region (AMNH 1909). *R. rattus* was observed from the Lake Albert plague foci in 1948 and was in the Lake Edward focus since 1936 (Davis 1953). The West Nile region of Uganda is comprised of ethnic group diversity distinct from the rest of Uganda. These ethnic groups (Nubi, Alur, Lugbara and others) retain close links with related groups over the borders with Sudan and DRC and, in many aspects, their ties to these groups are closer than their relationship with southern Ugandans (Leopold 2006). The West Nile region was historically maintained as a "closed area" from which outsiders were excluded, cash crops discouraged, and was a labor reserve that supplied workers for plantations in the southern part of Uganda (Leopold 2006). Although Uganda became a British protectorate in 1886, the West Nile region did not fall under this protectorate until 1914. Previously, it had been occupied by Belgian

troops from 1892 until King Leopold's death in 1909, then briefly incorporated into Sudan until 1914 (Leopold 2006). It is possible that the isolation of this region prevented rats from arriving until later in the 20th century (possibly from the East) and/or movement of trans-border populations led to the arrival of plague and rats to the West Nile region from the plague focus in the DRC to the west.

Overall, the major rodents responsible for plague outbreaks in Africa are thought to be *R. rattus*, *Mastomys* spp., *Arvicanthis* spp., and *Tatera* spp. *M. natalensis* and *A. niloticus* appear to be relatively resistant to infection and are, therefore, considered enzootic hosts (Politzer and Meyer 1961). *A. niloticus* ranges from Eastern to Western Africa and throughout the Nile valley to Egypt (Taylor 1984). The rodent is mainly herbivorous feeding on grasses and, unlike the majority of rodents, is diurnal. It is not normally a pest of stored grains, but will occasionally enter grass huts and grain stores (Fieldler 1988).

The major rodent-flea vectors responsible for *Y. pestis* transmission are *X. bransilensis* and *X. cheopis* (Politzer 1954, Gratz 1999). In northern Africa, rodent species of the genus *Gerbillus* and *R. rattus* are the primary reservoirs in the region with *X. cheopis, X. ramesis* and *X. nubica* being the principal flea vectors (Gratz 1999). The introduced (from South America) human flea, *Pulex irritans*, is also a likely transmitter of plague in some areas of Africa including the Lushoto region of northeast Tanzania (Gratz 1999, Laudisoit et al. 2007). The role of the European cat flea, *Ctenocephalides felis strongylus*, in the transmission of plague in East Africa is questionable. Historically, the cat flea was observed to be the most common flea found on humans and parasitizes dogs, cats, rodents, sheep and goats (Hopkins 1947). The cat flea (*P. irritans*) does not undergo "blocking" (the rapid proliferation of *Y. pestis* in the proventriculus, as with *X. cheopis*), which is thought to increase the vector capacity of *Y. pestis* transmission in fleas. Eisen et al. (2006) recently showed that *Oropsylla montana*, a flea of prairie dogs (*Cynomys* spp.) in the United States could transmit the bacteria in an "unblocked" state. *O. montana* does not, or rarely, becomes blocked, but was shown to be capable of transmitting infection for 4 days ("early phase transmission") and may remain infectious for up to 8 weeks. The mechanism for early phase transmission is still unknown, but may be from the regurgitation of previously consumed infectious bloodmeals. Early phase transmission in *C. felis* and *P. irritans* (Verjitski 1908 described in Eisen et al. 2006), was shown, but never effectively evaluated as important in the enzootic cycle of plague. These findings could cause a major re-thinking of the epizoology of historic and current plague outbreaks in the world.

The oriental rat flea, *X. cheopis*, seems to have made two separate invasions of East Africa, one from the north down the Nile Valley, independent of *R. rattus* and one from the East coast along with the arrival of *R. rattus* and transported inland with the rats (Hopkins 1947). The flea was reported in Ugandan areas absent of *R. rattus* (Hopkins 1947) and some authors suggest that the oriental rat flea, *X. cheopis*, coevolved with the Nile rat, *A. niloticus*, in northern Africa (Lewis 1967, Panagiotakopulu 2004). Many fleas including *X. cheopis* and *Ctenocephalides* spp. fleas are capable of growing inside indigenous huts. Politzer (1954) noted that in areas where grain is stored and hence the dust of grain, combined with constant temperature, little ventilation and relatively constant humidity young *X. cheopis* were very successful at breeding and developing.

Risk of exposure to infectious rat fleas are especially high in plague affected rural areas of certain developing countries that have large rat populations as a result of poverty and poor sanitation and food storage practices. Epidemics of bubonic plague are most likely to occur among residents of such areas when plague epizootics ("ratfalls") result in high mortality among rats and cause their fleas to seek other hosts, such as humans (Gage 1998). In West Nile communities, many individuals observed mortality of *R. rattus*, as a pre-cursor to the risk of human plague cases (Genesis Laboratories, unpublished data).

## MATERIALS AND METHODS

This study was performed in cooperation with the Ugandan Viral Research Institute, Ugandan Ministry of Health Plague Research Laboratory in Arua, Uganda. Four field sites were chosen in the Arua district, Vurra Sub-county, Opia and Ayanvu parishes. Each site was typical of a single indigenous homestead typical to the area and contained between 7-20 structures and was bordered by peridomestic area that included crops and habitat typical to the area. Sites were chosen through discussion with the homestead residents and based on negative reports of (1) recent die-offs of rodents, and (2) human plague cases in 2005.

Each homestead was sampled for both rodent and flea populations. Rodents were collected using live traps. If the structures were large (e.g., sleeping, storage or cooking areas), 3 Tomahawk traps (48.3 x 17.1 x 17.1 cm, Tomahawk, WI) were placed in and around each structure (2 inside and 1 outside). If the structure was small (e.g., latrine or washing areas), 2 Tomahawk traps were placed in and around each structure (1 inside and 1 outside). The peridomestic areas surrounding each village were sampled for rodents as well. The estimated center of the village was determined. Using a compass, Tomahawk traps were set on the north, south, east and west compass lines, starting at the edge of the peridomestic area, every ten meters for 100 m into the surrounding habitat. Each line therefore contained up to 11 traps (point 0 – point 10). In the event that trap lines encountered neighboring villages, large roads or habitat that prevented the continuation of the trap-line, they were shortened. Rodent trapping was performed for 3 nights before and after the rodent baiting period. Traps were checked twice a day (am and pm). When captured, each rodent was sedated using Halothane (Halocarbon Laboratories, River Edge, New Jersey, US). Fleas were collected from each rodent by combing them over a white plastic pan with a commercially available flea comb. The number of fleas was counted and recorded. The body weight of each rodent was recorded as well as measurements of body, tail and hind-foot and ear in some cases.

A maximum of 10 larger structures in each village were sampled for fleas. Free-living flea populations in structures were sampled using 3 different methods. The first method sampled free-living fleas within dwellings and employed (1) a modified Kilonzo type flea trap (based on Orachi-Orach 2002), which suspended a flashlight over a metal pan containing water (for collection of photosensitive fleas), and (2) an additional metal pan, containing water, placed in each structure to capture non-photosensitive fleas. Village residents were instructed to turn the flashlights on at night and to leave on throughout the night. Flea traps were used to collect traps for two nights. Fresh batteries were used for each night of collection. Secondly, each home was also evaluated for the presence of rodent burrows both inside and outside the structure. If rodent burrows were present, each burrow was sampled by a "burrow swabbing" technique. A ~15 x 15 cm piece of white flannel, attached to a plumbers snake was inserted into each burrow. The flannel was shaken vigorously as it was removed. Lastly, if clothing or bedding was present in the structures chosen for flea population evaluation, samples of clothing and bedding were removed and placed on a white sheet. Clothing was slowly unraveled and observed for the presence of fleas. Fleas were collected and saved for later evaluation.

The number of rodents collected from each site was totaled, and rodent flea index was defined as the total number of fleas collected from rodents divided by the number of rodents collected. Similarly, household and burrow swab flea indices were calculated by the number of fleas collected for each index divided by the number of traps placed in huts or burrow swabbed.

## RESULTS

On the 4 study locations, 37 commensal area dwellings were evaluated. A total of 94 rodents were collected, nearly equal amounts in both commensal and peridomestic areas (Table 1). Forty-five (47.8%) were captured in the commensal

**Table 1.** Species of rodents collected from commensal and peridomestic areas of all field sites.

| **Commensal (n=45) (Number Captured, % of Captures)** | **Peridomestic (n=49) (Number Captured, % of Captures)** |
|---|---|
| *Rattus rattus* (40, 88.9%) | *Arvicanthis niloticus* (28, 57.1%) |
| *Arvicanthis niloticus* (4, 8.9%) | *Mastomys natalensis* (6, 12.2%) |
| Shrew (species?) (1, 2.2%) | *Rattus rattus* (3, 6.1%) |
| | *Lophuromys flavopunctatus* (3, 6.1%) |
| | *Lophuromys sikapusi* (2, 4.1%) |
| | Gerbil? (Tatera?) (2, 4.1%) |
| | *Lemniscomys* spp. (1, 2.0%) |
| | Unknown species (4, 8.2%) |

representing three species of rodents, while 49 (52.2%) were captured in peridomestic areas, representing at least 8 different species of rodents. Of the rodents captured in commensal areas, the most commonly collected was the invasive *R. rattus* (88.9% of captures; Table 1). Nile rats, *A. niloticus*, comprised 4.1% of captures in commensal areas. In peridomestic areas, the most common rodent collected was *A. niloticus* comprising 57.1% of captures followed by *M. natalensis*, which comprised 12.2% of captures. Of the remaining rodents collected in peridomestic areas, two species represented capture rates of approximately 6% each and the remaining identified rodents <5% each (including *R. rattus*). Of the collected rodents, 8.2% were not identified in the field. Voucher specimens of these rodents were submitted to the Denver Museum of Nature and Science (Denver, Colorado, US). At the time of writing, positive identification had not been determined.

Rodents collected in commensal and peridomestic areas yielded flea indices of 2.3±1.3 and 2.0±0.7, respectively (Table 2). Considering only *R. rattus*, animals collected in commensal and peridomestic areas yielded identical flea indices of 1.7±1.2. Collections of free-living household fleas using traps yielded flea indices of 1.3±0.6 (light trap) and 0.5±0.4 (dark trap). A total of 66 burrows were swabbed for the collection of burrow-dwelling fleas. Fleas collected from swabbing yielded a flea index of 0.19±0.12 (Table 2).

## DISCUSSION

These data suggest that the most common rodent inhabiting these commensal areas are invasive *R. rattus*. No *M. natalensis* were captured in commensal areas supporting the findings of other authors, which indicate that *M. natalensis* is displaced by *R. rattus*. Less than 5% of peridomestic captures were *R. rattus*. The associated flea indices of both peridomestic and commensal rodents was not substantially great. Regarding household flea index obtained from the use of light and dark traps, Orochi-Orach (2002) reports that a household flea index 1.5 represents the need for urgent action to prevent an epidemic of plague. The average household flea index of lighted traps in our study was greater than 1.5 indicating an increased risk of transmission of disease to village residents.

In regards to control, the basic objective of plague prevention is to reduce the likelihood of humans being bitten by infected fleas. Control of rodents in the village setting is often complicated by the constant exposure of commensal rodents to peridomestic wild rodents. Even when rodents are effectively controlled in a village setting, wild rodents can often invade the village areas. Invasion of rodents often occurs after cropping and harvesting procedures and inhabitants should be warned of potential plague risk during these periods. Rodent control around a dwelling is important during this time, but more practically, storing food in rodent-proof containers and destruction of rodent harborage areas are good

**Table 2.** Mean Flea Index in Uganda.

| Collection | Mean Flea Index ± Standard Deviation |
|---|---|
| Commensal rodents[1] | 2.3±1.3 |
| Peridomestic rodents[2] | 2.0±0.7 |
| Commensal *R. rattus*[1] | 1.7±1.2 |
| All *R. rattus*[1, 2] | 1.7±1.2 |
| Household trap[3] | 1.3±0.6 (light trap); 0.5±0.4 (dark trap) |
| Commensal burrow swab[4] | 0.19±0.12 |

[1] Four field sites, n=37 dwellings evaluated.
[2] Starting at edge of commensal area.
[3] Total number of fleas captured / number of trap-nights.
[4] Total number of fleas captured / number of burrows swabbed.

ways to prevent invasion. Storing grain in the ceilings of dwellings will allow a food source for rodents and bring them into close contact with humans. Unkempt dwellings with an accumulation of solid food wastes promote rodent breeding and the distance between the human dwelling and the refuse pile was a risk factor for rat infestation (Olascha et al. 1994, Makudi et al. 1999). Conversion of land into agriculture increases the habitat for sylvatic rodents and potential of interaction with peridomestic rodents (Makudi et al. 1999). Clearing brush from around a dwelling can create a hindrance to immigrating rodents. Other preventative measures include not sleeping on the floor and occasionally pouring boiling water on the floor of dwelling for sanitation. Fleas typically are associated with dirty or dirt floors, and this method could be especially useful during epidemic periods to kill fleas in dwellings. The use of cats to control plague vectors is questionable. Cats are helpful at controlling rodents, but should be avoided if sick because of potential spread of pneumonic plague (Kilonzo 1999). In Zimbabwe, epidemiological analysis after a human plague outbreak indicated that the presence of a sick cat was a risk factor in contracting the disease (Manungo et al. 1998).

Plague control targeting rodents elsewhere in Africa has focused on dusting of human dwellings and rodent burrows with powdered insecticide, rodent trapping, and education in Namibia (Shangula 1998), DDT powder followed by anticoagulant bait in Botswana (Kumaresan et al. 1991), trapping in Tanzania, Nigeria, and Madagascar, bounty schemes and rodenticide use in Tanzania as well as the burning of dwellings while villagers killed escaping rodents with clubs (Kilonzo 1994, Kilonzo 1999). Recently in Madagascar, deltamethrin and diazinon dust were evaluated to control fleas on rats (primarily *R. rattus*). Dust was placed at burrow openings. Flea suppression was obtained during the first month, but each dusting was unsuccessful at maintaining residual control of fleas on the rodent host (Rotovonjato and Duchemin 2001). Often in Africa, plague control is performed reactively, not proactively (Govere and Dürrheim 1999), often due to lack of resources for surveillance. In Tanzania, governmental financial constraints on plague control activities in some years are believed to have led to the outbreaks of plague in following years (Kilonzo and Komba 1993).

Regardless of the method used to control rodents and fleas, it is imperative that flea control be performed before, or concurrently with, rodent control practices. Lethally controlling rodents without first controlling fleas could create an increased risk of plague to humans as many fleas are released into the environment after widespread rodent deaths occur (Gage et al. 1995). An example of this occurred in Tanzania in which an epidemic of human plague was reported to be preceded by rodent control operations indiscriminately using rodenticides, thereby causing fleas to be released into the environment and, subsequently, parasitizing humans (Kilonzo 1994). Likewise, a case control study in Peru investigating the risk factors of bubonic plague found a strong association with indiscriminate rat extermination (Odds Ratio, OR=71.4). Other risk factors determined were: the tendency to sleep on the floor (OR=16), the presence of rubbish inside and outside of the dwelling (OR=8.89), the death of guinea pigs, the increase of flea and rodent populations, and the storage of food inside the dwelling (Angulo et al. 1997).

Regardless of how and when *R. rattus* arrived in the West Nile region of Uganda, the presence of the invasive rodent appears to now be critical in the epizootic cycle of plague. The rodent arrived into an area that had an already established, or soon to be established, a foci of plague, likely displaced the resident commensal rodent populations and took its place near humans, as it has done efficiently throughout the world. Unfortunately, its presence has been detrimental to the people of Northwest Uganda causing increased disease burden of plague. Additional research is needed to clearly understand the zoonotic cycle of plague in this region as well as methods to prevent and control the disease.

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## LITERATURE CITED

AKIEV, A. K. 1982. Epidemiology and incidence of plague in the world, 1958-1979. Bulletin of the World Health Organization 60:165-169.

ALPIN, K. P., T. CHESSER, AND J. T. HAVE. 2003. Evolutionary biology of the genus *Rattus*: profile of an archetypal rodent pest. *In* G. R. Singleton, L. A.

Hinds. C. J. Krebs, and D. M. Sprat, editors. Rats, mice and people: rodent biology and management. Austrailan Centre for International Agriculture Research, Canberra.

ANONYMOUS. 1939. Uganda: annual report of the Medical Department, 1938. Medical Department, Entebbe.

AMERICAN MUSEUM OF NATURAL HISTORY (AMNH). 1909. Online database of field notes collected by the American Museum of Natural History Congo Expedition 1909-1915, by H. Lang and J. P. Chapin. Downloaded 2007, http://diglib1.amnh.org.

ANGULO, O. C., E. VARGAS, AND J. BENITES. 1997. Case-control study to determine the associated risk factors of the presence of bubonic plague in Tunad, Hualabamba and El Gigante villages in San Bernardino District-San Pablo-Cajamarca, Peru. Journal of Clinical Epidemiology. Abstract. 50(1), Supplement 1:27S.

AVERY, D. M. 1985. The dispersal of brown rats *Rattus norvegicus* and new specimens from 19th century Capetown. Mammalia 49:573-576.

BARRETT, R.E. 1933. Epidemiological observations on plague in the Lango district of Uganda. East African Medical Journal 10:160-180.

CONRAD, L. I. 1982. TĀ'ŪN and WABĀ' conceptions of plague and pestilence in early Islam. Journal of the Economic and Social History of the Orient 25:268-307.

DAVIS, D. H. S. 1953. Plague in Africa 1935-1949 a survey of wild rodent in African territories. World Health Organization 9:665-700.

DAVIS, D. H. S., AND B. M. FAGAN. 1962. Sub-fossil house rats (*Rattus rattus*) form Iron Age sites in Northern Rhodesia. Zoological Society of Southern Africa 3:13-15.

DAVIS, D. H. S., R. B. HEISCH, D. MCNEIL, AND K. F. MEYER. 1968. Serological survey of plague in rodents and other small mammals in Kenya. Transactions of the Royal Society of Tropical Medicine and Hygiene 62:838-861.

EISEN, R J., S. W. BEARDEN, A. P. WILDER, J. A. MONTENIERI, M. F. ANTOLIN, AND K. L. GAGE. 2006. Early-phase transmission of Yersinia pestis by unblocked fleas as a mechanism explaining rapidly spreading plague epizootics. Proceedings of the National Academy of Sciences 103:15300-15385.

FIELDER, L. A. 1988. Rodent problems in Africa. *In* Rodent pest management. CRC Press, Inc. Boca Raton, Florida, USA.

FIELDER, L.A. 1994. Rodent pest management in Eastern Africa. FAO Plant Production and Protection Paper 123.

GAGE, K. L. 1998. Plague. *In* W. J. Hausler, and M. Sussman, editors. Microbiology and microbial infections volume 3 bacterial infections. Oxford University Press, New York, USA.

GAGE, K. L., R. S. OSTFELD, AND J. G. OLSON. 1995. Nonviral vector-borne zoonoses associated with mammals in the United States. Journal of Mammology 76:695-715.

GOVERE, J., AND D. N. DÜRRHEIM. 1999. Plague surveillance in South Africa. South African Medical Journal 89:570.

*(a)* GRATZ, N. G. 1999. Control of plague transmission. *In* Plague manual epidemiology, distribution, surveillance and control. World Health Organization publication WHO/CDS/CSR/EDC/99.2.

GUIYOULE, A., F. GRIMONT, I. ITEMAN, P. A. D. GRIMONT, M. LEFÈVRE, AND E. CARNIEL. 1994. Plague pandemics investigated by ribotyping of Yersina pestis strains. Journal of Clinical Microbiology 32:634-641.

HOPKINS, G. H. E. 1947. Annotated and illustrated keys to the known fleas of East Africa. The Ugandan Journal 11:133-190.

HOPKINS, G. H. E. 1949. Report on rats, fleas and plague in Uganda. Government Printer of Uganda.

JANSSENS, P. G., AND S. R. PATTYN. 1994. Plague in Zaire. Verh K Acad Geneeskd Belg 56:281-361.

KEYES. D. 1999. Catastrophe an investigation into the origins of the modern world. Ballantine Books, New York, USA.

KILONZO, B. S. 1976. A survey of rodents and their flea ectoparasites in North-Eastern Tanzania. East African Journal of Medical Research 3:117-126.

KILONZO, B. S. 1994. Importance of intersectoral co-ordination in the control of communicable diseases, with special reference to plague in Tanzania. Central African Journal of Medicine 40:186-192.

KILONZO, B. S. 1999. Plague epidemiology and control in Eastern and Southern Africa during the period 1978 to 1997. Central African Journal of Medicine 45:70-76.

KILONZO, B. S., AND E. K. KOMBA. 1993. The current epidemiology and control of trypanosomiasis and other zoonoses in Tanzania. Central African Journal of Medicine 39:10-20.

KRASNOV, B. R., AND G. I. SHENBROT. 2002. Coevolutionary events in the history of association between jeroboas (rodentia: dipodidae) and their flea parasites. Israel Journal of Zoology 48:331-350.

KUMARESAN, J. A., J. B. GROVA, P. K. MMATLI, AND E. D. MAGANU. 1991. An experience in the control of plague in Botswana. Tropical Doctor 10:142-146.

LAUDISOIT, A., H. LEIRS, R. H. MAKUNDI, S. V. DONGEN, S. DAVIS, S. NEERINCKX, J. DECKERS, AND R. LIBOIS. 2007. Plague and the human flea, Tanzania. Emerging Infectious Diseases 13:687-693.

LEIRS, H., R. VERHAGEN, W. VERHEYEN, P. MWANJABE, AND T. MBISE. 1996. Forcasting rodent outbreaks in Africa: an ecological basis for *Mastomys* control in Africa. Journal of Applied Ecology 33:937-943.

LEOPOLD, M. 2006. Legacies of slavery in Northwest Uganda: the story of the one-elevens Africa 76:180-199.

LEWIS, R. E. 1967. The fleas (siphonaptera) of Egypt. An illustrated and annotated key. The Journal of Parasitology 53:863-885.

MAKUNDI, R H., N. O. OGUGE, AND P. S. MWANJABE. 1999. Rodent pest management in East Africa-an ecological approach. *In* G. R. Singleton, L. A. Hinds, H. Leirs, and Zhibin Zang, editors. Ecologically-based rodent management. ACIAR Monograph No. 59.

MANUNGO, P, D. E. PETERSON, C. H. TODD, N. MTHAMO, AND B. PAZVAKAVAMBA. 1998. Risk factors for contracting plague in Nkayi district, Zimbabwe. Central African Journal of Medicine 44:173-6.

MCEVEDY, C. 1988. The bubonic plague. Scientific American. 258:188-123.

MSANGI, A. S. 1975. The surveillance of rodent populations in East Africa in relation to plague Endemicity. University Science Journal 1:6-20.

NOWAK, R. M. 1991. Order rodentia. *In* Walker's Mammals of the world. 5th Edition, volume 1. Johns Hopkins University Press, Baltimore, Maryland, USA.

OGEN-ODOI, A. A. 1999. Ecological factors influencing the epidemiology of plague in Uganda. Research proposal, Department of Zoology, Makerere University, Kampala, Uganda.

OLASEHA, I. O., M. K. C. SRIDHAR, P. C. OBIAKO, AND A. OLADAPP. 1994. Rat infestations in urban and rural areas in Nigeria: public health implications. The Journal of the Royal Society of Health 12:300-303.

OROCHI-ORACH, S. 2002. Plague outbreaks the gender and age perspective in Okoro county, Nebbi district, Uganda. Agency for Accelerated Regional Development.

PANAGIOTAKPULU, E. 2004. Pharaonic Egypt and the origins of plague. Journal of Biogeography 31:269-275.

POLITZER, R. 1954. History and present distribution of plague. Page 11 *in* Plague. World Health Organization, Geneva.

POLITZER, R., AND K. F. MEYER. 1961. The ecology of plague. *In* J. M. May, editor. Studies in disease ecology. Hafner Publishing Co., New York, USA.

PLUG, I., N. J. DIPPENAAR, AND E. O. M. HANISCH. 1979. Evidence of *Rattus rattus* (house rat) from Point Drift, an Iron Age site in the Northern Transvaal. South African Journal of Science 75:82.

ROTOVONJATO, J., AND J. B. DUCHEMIN. 2001. Evaluation de l'effet du Knox-Out microencapsulé V.O. 240 et de la K-othrine poudre sur les puces des rats de deux villages de region de Betafo. Arch Inst Pastuer de Madagascar 67:46-48.

SHANGULA, K. 1998. Successful plague control in Namibia. South African Medical Journal 88:1428-1430.

SHENBROT, G., AND B. KRASNOV. 1999. Geographic variation in the role of gerbils and jirds (gerbillinae) in rodent communities across the Great Palaeartic Desert Belt. *In* C. Denys, L. Granjon, and A. Poulet, editors. Proceedings of the 8th International Symposium on African Small Mammals. IRD Editions, Paris, France.

STAPLES, J. E., J. KOOL, AND P. MEAD. 2004. Characteristics of plague cases – Uganda, 1989-2003. Submitted abstract, CDC EIS Conference.

TAYLOR, K. D. 1984. Vertebrate pest problems in Africa. Proceedings of a Conference on the Organization and Practice of Vertebrate Pest Control. ICI Plant Protection Division, Haslemer, England.

TIKHOMIROV, E. 1999. Epidemiology and distribution of plague. Plague manual epidemiology, distribution, surveillance and control. World Health Organization WHO/CDS/CSR/EDC/99.2.

TWIGG, G. I. 1978. The role of rodents in plague dissemination: a worldwide review. Mammal Review 8:77-110.

VOIGT, E. A., AND A. VON DEN DRIESCH. 1984. Preliminary report on the faunal assemblage from Ndondondwane, Natal. Annals of the Natal Museum 26:95-104.

WORLD HEALTH ORGANIZATION. 2000. Chapter 3: Plague. WHO Report on global surveillance of epidemic-prone infectious diseases. WHO/CDS/ISR/2000.1.

WORLD HEALTH ORGANIZATION. 2003. Human plague in 2000 and 2001. Weekly Epidemiological Record 16, April 18, 2003.

WORLD HEALTH ORGANIZATION. 2004. Human plague in 2002 and 2003. Weekly Epidemiological Record 33:310-308.

ZIETZ, B. P., AND H. DUNKELBURG. 2004. The history of the plague and the research on the causative agent *Yersinia pestis*. International Journal of Hygiene and Environmental Health 207:165-178.

# ERADICATING HOUSE MICE FROM ISLANDS: SUCCESSES, FAILURES AND THE WAY FORWARD

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***Abstract:*** The house mouse (*Mus musculus*) has been spread throughout the world by the actions of humans. It causes severe impacts to native ecosystems, especially in areas where there are no native mammals. It is possible to eradicate mice from islands but they are harder to eradicate than rats. A review of reported eradication attempts found that 17 attempts on 45 islands worldwide failed; a failure rate of 38%. The effect of operational factors on eradication success was examined, but no significant model was formed. Brodifacoum is the most widely used toxicant and has a 49% success rate. Mouse eradications should be attempted wherever possible and recommendations to help increase the success of a house mouse eradication attempt are given.

***Key words:*** brodifacoum, eradication, house mouse, invasive species, island conservation, *Mus musculus*, rodenticide.

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## INTRODUCTION

The house mouse (*Mus musculus*) originated in the north of India around 900,000 years ago (Boursot et al. 1996). The species then spread in several directions, radiating to form three distinct sub-species (*M. m. domesticus, M. m. musculus* and *M. m. castaneus*) with distinct ranges (Boursot et al. 1993, Boursot et al. 1996). All sub-species show a high level of commensal behaviour (Boursot et al. 1996, Berry and Scriven 2005) but they are also able to survive away from human settlements (Berry and Scriven 2005, Ruscoe and Murphy 2005). The commensal behaviour of house mice means they have been spread throughout the world by humans, and house mice are present on all continents and many islands from the sub-Antarctic to the tropics (Berry and Scriven 2005, Ruscoe and Murphy 2005, Wanless et al. 2007, Witmer et al. 2007). The effect of introduced, invasive house mice has often been overshadowed by invasive rats (*Rattus* spp.), however (e.g., Atkinson 1985), especially where they co-exist and mice are dominated by rats (Caut et al. 2007). Non-commensal populations of house mice can have severe negative impacts on native ecosystems, especially in areas where the native biota evolved in the absence of mammals (Courchamp et al. 2003), and house mice have been recorded as damaging populations of invertebrates (Rowe-Rowe et al. 1989, Miller and Webb, 2001), lizards (Newman 1994), birds (Jones et al. 2003, Wanless et al. 2007) and seed production in forests (Wilson et al. 2007). Eradication of invasive rodents is an important management tool to redress their negative impacts and a recent review recorded that introduced house mice have been eradicated from 30 islands worldwide, using a number of different methods. Despite this progress, seven attempts failed which is a 19% failure rate, compared to a 5% failure rate for Norway rats (*Rattus norvegicus*) (Howald et al. 2007). Is there a reason that introduced mouse populations are harder to eradicate from islands than introduced rat populations? In order to answer this question, we compiled, reviewed and analysed a database of all known mouse eradication attempts. The database was compiled from the published literature, "grey" literature, and through conversations with researchers and managers involved in introduced house mouse eradication attempts (see Appendix).

## ISLAND MOUSE ERADICATIONS

The first reported mouse eradication took place on Flatey Island in Iceland in 1971 (Moors 1992). Since then, there have been over 50 other attempts worldwide from Rasa Island in the Gulf of California (Tershy et al. 2002) to Enderby Island in the sub-Antarctic (Torr 2002). Different toxicants

and broadcasting methods have been used in conjunction with trapping in some cases. Eradication attempts have taken place on 51 islands ranging in size from 0.7 ha Crusoe Island in New Zealand (Lee 1999) to 800 ha St. Paul Island in the French Sub-Antarctic (Micol and Jouventin 2002). Successes and failures have occurred across the full range of island sizes (see Appendix). Two eradication attempts were stopped before completion for operational reasons and six are yet to be confirmed. Eradication of house mice was achieved on 28 of 45 islands that the result is known for. However, sometimes it took more than one attempt. On Mokoia Island, New Zealand the first two operations failed but the third attempt was successful. All four operations on Limestone Island, New Zealand have failed. This gives a failure rate of 38% which is higher than reported by Howald and others (2007) and much higher than failures reported for rat species. A total of over 3,600 ha of island habitat worldwide has been cleared of mice.

We categorised each house mouse eradication attempt by four operationally defined factors which might affect the likelihood of successful eradication (Table 1). In order to identify which (if any) of these factors most influence eradication success or failure a logistic general linear model was fitted with success/failure as the response factor and details of the eradication attempt entered as explanatory variables. No significant model was formed with any combination of explanatory variables meaning there is no evidence that success or failure of mouse eradications to date has been consistently caused by any of these operational factors. Nonetheless we report success and failure rates relative to each factor.

**Toxicants**

Nearly all recorded mouse eradication attempts used some form of anticoagulant toxicant. These compounds are used in eradication attempts worldwide (Eason et al. 2002, Hoare and Hare, 2006) and act by inhibiting the production of clotting factors within the animal normally leading to death by internal haemorrhage within 10 days (O'Connor and Booth 2001). Seven toxicants have been used in mouse eradication attempts; three first-generation anticoagulants (diphacinone, pindone and warfarin) were used as the main toxicant in six attempts, three second-generation anticoagulants (brodifacoum, bromadiolone and flocoumafen) were used as the main toxicant in 49 attempts and an acute toxicant (1080) in one. Five attempts used multiple toxicants and two attempts followed up poisoning with trapping. Brodifacoum was used as the main or secondary toxicant in 80% of mouse eradication attempts (including multiple attempts on the same island), 49% of which were successful (45 attempts, 22 successful). Other toxicants have a higher success rate but the sample size is much lower. A single eradication attempt using 1080 (Varanus Island, Australia, 1993) is likely to have failed because it has been shown that mice can detect the presence of 1080 in baits (O'Connor et al. 2005).

**Bait Delivery**

Three main methods of bait delivery have been used in mouse eradication attempts. The method chosen depends on island topography, non-target issues, economics and the habitat on the island (Howald et al. 2007). Information is scarce on the earliest recorded mouse eradication attempt (Flatey

**Table 1.** Factors investigated in analysis of eradication attempts.

| Factor | Description |
|---|---|
| Island area | Size of the island in hectares |
| Bait application method | Aerial, bait station or hand spreading |
| Toxicant (generation: 1st or 2nd generation anticoagulant) | Diphacinone (1), pindone (1), warfarin (1), brodifacoum (2), bromadiolone (2) or flocoumafen (2) |
| Other introduced mammals | Competitors, predators or no direct effect |

Island, Iceland, 1971) but it has been assumed that bait stations were used.

1. Bait stations were used as the main method of bait delivery in 30 out of 56 eradication attempts (including multiple attempts on the same island). They were also used to supplement aerial delivery in two attempts. The grids used for bait station delivery varied in size from 10 m to 50 m; 20 m to 25 m being the most common spacing used. Bait station grids are normally maintained for 1-2 years (Thomas and Taylor 2002) but some attempts went on for much longer. Bait stations were first placed on 37 ha Limestone Island, New Zealand in 1999 and have been regularly serviced for over 6 years (J. Craw, Auckland Regional Council, New Zealand, personal communication) but mice are still present, despite three aerial attempts and one ground-based attempt, and prolonged periods of non-detection (C. Mitchell, Limestone Island Ranger, New Zealand, personal communication). Bait stations are relatively labour intensive and track maintenance can damage island habitat; particularly with smaller grid spacing; but if the support required to service bait stations is available this is a relatively effective method with 48% of eradication attempts succeeding. The largest island successfully cleared of mice using this method was 253 ha Flat Island in Mauritius using a 25 m by 25 m grid (Bell 2002).

2. Hand broadcasting of baits was used in two eradication attempts; both run by French teams, and where one attempt was successful and the other failed. Fajou Island in Guadaloupe is the largest island (120 ha) where mouse and rat eradication was attempted using this method and poisoning in this instance was supplemented by trapping (M. Pascal, National Institute for Agricultural Research, France, personal communication). A recent visit to the island found mice present at low numbers but the reason for eradication failure is unclear (M. Pascal, personal communication). Hand broadcast is a valuable method to consider when aerial broadcast is not possible and when the continued support needed to maintain a network of bait stations is unavailable. Hand broadcasting of baits has been used to supplement a number of bait station and aerial operations to ensure bait reaches all areas of islands (Stephenson et al. 1999, Merton et al. 2002).

3. Aerial broadcast of bait using helicopters is becoming more common and the preferred method of bait delivery for introduced rodent eradications (Towns and Broome 2003). This technique has been used in 25 mouse eradication attempts around the world. In some cases aerial operations have been supplemented by hand broadcast or bait stations, but the majority of attempts rely solely on bait distributed by helicopter. Forty eight percent of eradication attempts using aerial broadcast have been successful. The amount of bait distributed onto the island and the number of bait drops varies. This information is not always available but the mean quantity of bait used in 16 operations was 15.3 kg $ha^{-1}$ (range 10-39 kg $ha^{-1}$). The number of drops varies between one and three. The highest bait density was used on Frégate Island in the Seychelles where the presence of crabs meant a large amount of bait had to be used (Merton et al. 2002). The flight paths of the helicopters are crucial to ensuring eradication success. Overlapping flight paths and second drops at right angles to the first are good methods of ensuring complete coverage of the island. Modern global positioning system (GPS) satellite technology allows helicopter pilots to plot locations and flight paths very accurately (Lavoie et al. 2007). Five recent eradication attempts in New Zealand had bait distributed by helicopter, but we are awaiting confirmation of success. We did not model the amount of bait used, or number of bait drops, but these operational factors, which are island-specific, may affect the outcome of eradication attempts.

**Other Mammal Species**

Populations of mice are significantly affected by the presence of other invasive mammal species (Innes et al. 1995, Choquenot and Ruscoe 2000). There have been a number of reported instances where mice have increased in number once rats have been eradicated or brought to low numbers (Caut et al. 2007). The presence of other mammal species may alter the behaviour of mice and make them less likely to come into contact with bait, leading to eradication failure (Innes et al. 1995). Where possible the presence of other introduced mammal species has been recorded on each island where an eradication was attempted. Twenty-seven eradications were attempted in the presence of other mammal species and 13 of these failed (48%). The mammals present were then divided into three categories – competitors (rat species); predators (cats [*Felis catus*], stoats [*Mustela erminea*], weasels[*Mustela* spp.]) and no direct effect (rabbits [*Oryctolagus cuniculus*] and brushtail possums [*Trichosurus vulpecula*]). Interactions between rats

and mice are complex and poorly understood and there is likely to be an element of both competition and predation (Caut et al. 2007). Rabbits and possums have no direct impact on mouse populations but can eat bait and therefore stop mice accessing it. On Motuihe, New Zealand, high rabbit numbers may have reduced the amount of bait available to rats and mice but the eradication was still successful (Veitch 2002). Dividing the mammal species into different categories had no effect on the model.

## WHY DO MOUSE ERADICATION ATTEMPTS FAIL?

In order for an eradication to succeed every house mouse on an island must have access to the toxicant. At the most basic level poor operational implementation during the baiting campaign may lead to areas of the island being missed by bait. A retrospective assessment of operational implementation effectiveness could not be included as a variable here due to its subjective nature. However, one of the main reasons for mouse eradication attempts failing could be gaps in poison coverage. An eradication attempt on St. Paul Island in the sub-Antarctic failed because a malfunction in the bait spreader led to gaps in coverage (Micol and Jouventin 2002). Similar problems with operational implementation may have occurred in other eradications and not been reported. In these cases, reasons for failure are clear and relatively simple to rectify in subsequent attempts. For some eradications, however, reasons for failure may be more complex and harder to demonstrate and resolve. Recently it has become apparent that even aerial operations using helicopters guided by GPS may leave gaps in poison coverage (Josh Kemp, Department of Conservation, New Zealand, unpublished data). Possibly some aspect of mouse behaviour means that in a number of cases some individuals are not being poisoned. These animals may not come into contact with the bait; they may find bait but not eat it (i.e., have a cereal aversion, Humphries et al. 2000) or they may have a level of toxicant resistance allowing them to survive eating the bait (e.g., mice on Lord Howe Islands are resistant to warfarin following ongoing control since 1986 (Billing 2000)). Research in laboratory situations has shown critical differences in spatial and social behaviours between wild and laboratory house mice ( Augustsson and Meyerson 2004, Augustsson et al. 2005) and between different chromosomal strains of wild house mice (Ganem and Searle 1996). Behavioural differences at the subspecies level may also contribute to some of the failures.

Introduced house mice are physiologically very different from invasive rats, and able to sustain island densities orders of magnitude higher. What now seems a straightforward eradication for invasive rats, may still remain a challenge for introduced mice (Howald et al. 2007). Despite this, eradicating mice should always be attempted provided sufficient information is gathered prior to eradication to ensure correct operational implementation (i.e., bait delivery method and toxicant amounts).

We were unable to create a model predicting success or failure of a mouse eradication attempt based on operational factors. Some operational factors appear to aid success, even if this is not statistically significant. Some observations from the database are as follows:

- Following an aerial bait operation with hand spreading of poison in at risk areas or use of bait station may increase eradication success.
- Hand spreading bait in conjunction with bait stations may lead to an increased chance of success.
- Multiple toxicants may result in success. Five successful eradication attempts combined brodifacoum with another toxicant.

Bait stations spaced at around 20 m apart had the best chance of success.

## FUTURE RESEARCH

Data on island house mouse populations are scarce, and only a few islands have been studied intensively (e.g., Marion Island (Avenant and Smith 2004, Ferraira et al. 2006, van Vuuren and Chown 2007) and Allports Island (Murphy 1989)). Basic information about home range sizes, ranging behaviour and densities on islands remain largely unknown, especially during critical winter months where on temperate islands mouse impacts may be greatest (Wanless et al. 2007). The effect of different habitat types on eradication attempt success is also unknown. Mice living in complex habitats with ample food may have small home ranges and not come into contact with bait. The response of mouse populations to poisoning has not been investigated on islands and nothing is known about how mouse populations re-colonise areas following a failed eradication attempt. Genetic samples should be taken prior to any eradication attempt to allow failed eradications to be

distinguished from re-invasions (Abdelkrim et al. 2007). Although eradication failure is never a desirable outcome, much knowledge can still be gained from reflecting on causes of an eradication failure.

Current research on introduced house mice at The University of Auckland and elsewhere is obtaining accurate density estimates using capture-mark-recapture techniques, investigating home range size and ranging behaviour using radio-tracking and other tracking techniques, and then monitoring the response of a mouse population to poison during an experimental eradication attempt. Recent laboratory work showed that most mice died eight days after first being fed bait, while a few survived for up to 21 days (G. Morriss, unpublished data). Trapping on Adele Island in New Zealand eight days after the first poison drop failed to detect any mice over 330 trap-nights and 40 tracking-nights across the entire island (J. MacKay, unpublished data). Toxicant resistance on islands where long-term poison campaigns are taking place may also be an issue (Billing 2000) and could explain why mice are still present despite repeated attempts to eradicate them.

## CONCLUSION AND RECOMMENDATIONS

Introduced house mouse eradication is an important conservation tool that should be used in order to mitigate the negative effects of introduced mouse populations on islands. Thirty eight percent of mouse eradication attempts on islands worldwide have failed, but we were unable to find a consistent simple operational explanation for these failures. Eradications should be attempted provided sufficient planning and preparation has taken place to rule out failure due to operational errors, or factors that can be controlled for. Brodifacoum is the most widely used toxicant. Poison bait has been distributed using bait stations, hand broadcast and aerial operations; each of these techniques has resulted in some successes and each technique has its merits. The presence of other mammal species on an island may affect the outcome of a mouse eradication attempt, but we detected no definitive affect on success of eradication. Further research is needed into mouse populations on islands to investigate what aspects of mouse ecology and behaviour lead to eradication failures.

Mouse eradications should always be attempted if adequate distribution of bait is feasible. However, eradications must be well planned to avoid failure. Factors to consider in order to maximise the likelihood of success include:

- Will the chosen poisoning method allow every mouse on the island access to poison?
- Take genetic samples prior to the eradication attempt. This allows the distinction to be made between eradication failure and a re-invasion and also can be used to determine sub-species.
- Consider the effects of other mammals. Will they prevent mice accessing poison?
- Will the mice eat the bait? Consider bait trials to check for poison palatability and cereal aversion.
- Are there areas which may require extra poison? Dense grassland can support very high numbers of mice and may require more poison than forest areas.

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## LITERATURE CITED

ABDELKRIM, J., M. PASCAL, AND S. SAMADI. 2007. Establishing causes of eradication failure based on genetics: case study of ship rat eradication in Ste. Anne Archpelago. Conservation Biology 21:719-730.

ATKINSON, I. A. E. 1985. The spread of commensal species of *Rattus* to oceanic islands and their effects on island avifaunas. Pages 35-80 *in* P. J. Moors, editor. Conservation of island birds. Cambridge: International Council for Bird Preservation Technical Publication No. 3.

AUGUSTSSON, H., K. DAHLBORN, AND B. J. MEYERSON. 2005. Exploration and risk assessment in female wild house mice (*Mus mus musculus*) and two laboratory strains. Physiology & Behaviour 84:265-277.

AUGUSTSSON, H., AND B. J. MEYERSON. 2004. Exploration and risk assessment: a comparative study of male house mice (*Mus musculus musculus*) and two laboratory strains. Physiology & Behaviour 81:685-698.

AVENANT, N. L., AND V. R. SMITH. 2004. Seasonal changes in age class structure and reproductive status of house mice on Marion Island (sub-Antarctic). Polar Biology 27:99-111.

BARWELL, E. 2002. Relative densities and diet of mice in four habitats at Tawharanui Regional Park. MSc, Auckland, New Zealand.

BELL, B. D. 2002. The eradication of alien mammals from five offshore islands, Mauritius, Indian Ocean. *In* C. R. Veitch and M. N. Clout, editors. Turning

the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

BERRY, R. J., AND P.N. SCRIVEN. 2005. The house mouse: a model and motor for evolutionary understanding. Biological Journal of the Linnaean Society 84:335-347.

BILLING, J. 2000. The control of introduced *Rattus rattus* on Lord Howe Island. II. The status of warfarin resistance in rats and mice. Wildlife Research 27:659-661.

BOURSOT, P., J. C. AUFFRAY, J. BRITTON-DAVIDIAN , AND F. BONHOMME . 1993. The evolution of house mice. Annual Review of Ecology and Systematics 24:119-152.

BOURSOT, P., W. DIN, R. ANAND, D. DARVICHE , B. DOD, F. VON DEIMLING, G. P. TALWAR, F. BONHOMME. 1996. Origin and radiation of the house mouse: mitochondrial DNA phylogeny. Journal of Evolutionary Biology 9:391-415.

CAUT, S., J. CASANOVA, E. VIRGOS, J. LOZANO, G. WITMER, AND F. COURCHAMP. 2007. Rats dying for mice: modeling the competitor release effect. Austral Ecology 32:858-868.

CHOQUENOT, D., AND W. A. RUSCOE. 2000. Mouse population eruptions in New Zealand forests: the role of population density and seedfall. Journal of Animal Ecology 69:1058-1070.

COURCHAMP, F., J. L. CHAPUIS, AND M. PASCAL. 2003. Mammal invaders on islands: impact, control and control impact. Biological Reviews 78:347-383.

EASON, C. T., E. C. MURPHY, G. R. G. WRIGHT, E. B. SPURR. 2002. Assessment of risks of brodifacoum to non-target birds and mammals in New Zealand. Ecotoxicology 11:35-48.

FERREIRA, S. M., R. J. VAN AARDE, AND T. D. WASSENAAR. 2006. Demographic responses of house mice to density and temperature on sub-Antarctic Marion Island. Polar Biology 30:83-94.

GANEM, G., AND J. B. SEARLE. 1996. Behavioural discrimination among chromosomal races of the house mouse (*Mus musculus domesticus*). Journal of Evolutionary Biology 9:817-830.

HOARE, J. M., AND K. M. HARE. 2006. The impact of brodifacoum on non-target wildlife: gaps in knowledge. New Zealand Journal of Ecology 30:157-167.

HOWALD, G., C. J. DONLAN, J. P. GALVÁN, J. C. RUSSELL, J. PARKES, A. SAMANIEGO, Y. WANG, D. VEITCH, P. GENOVESI, M. PASCAL, A. SAUNDERS, AND B. R. TERSHY. 2007. Invasive rodent eradication on islands. Conservation Biology 21:1258-1268.

HUMPHRIES, R. E., R. M. SIBLY, AND A. P. MEEHAN. 2000. Cereal aversion in behaviourally resistant house mice in Birmingham, UK. Applied Animal Behaviour Science 66:323-333.

INNES, J. G., B. WARBURTON, D. WILLIAMS, H. SPEED, AND P. RADFIELD. 1995. Large-scale poisoning of ship rats (*Rattus rattus*) in indigenous forests of the North Island, New Zealand. New Zealand Journal of Ecology 19:5-17.

JONES, A. G., S. L. CHOWN, AND K. J. GASTON. 2003. Introduced house mice as a conservation concern on Gough Island. Biodiversity and Conservation 12:2107-21109.

LAVOIE, C., C. J. DONLAN, K. CAMPBELL, F. CRUZ, AND G. V. CARRION. 2007. Geographic tools for eradication programs of insular non-native mammals. Biological Invasions 9:139-148.

LEE, M. 1999. Biota of seven islets off Waiheke Island, Inner Hauraki Gulf. Tane 37:99-136.

MERTON, D., G. CLIMO, V. LABOUDALLON, S. ROBERT, AND C. MANDER. 2002. Alien mammal eradication and quarantine on inhabited islands in the Seychelles. *In* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

MICOL, T., AND P. JOUVENTIN. 2002. Eradication of rats and rabbits from Saint-Paul Island, French Southern Territories. Pages 199-205 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. Gland, Switzerland: IUCN.

MILLER, A. P., AND P. I. WEBB. 2001. Diet of mice (*Mus musculus*) on coastal sand dunes, Otago, New Zealand. New Zealand Journal of Zoology 28:49-55.

MOORS, P. J., I. A. E. ATKINSON, AND G. H. SHERLEY. 1992. Reducing the rat threat to island birds. Bird Conservation International 2:93-114.

MORRISS, G. A. 2007. Susceptibility of Rangitoto and Motutapu island house mice to 20R brodifacoum baits. Report to the Department of Conservation by Landcare Research.

MURPHY, E. C. 1989. The demography of an island and mainland population of house mice in the Marlborough Sounds, New Zealand. Unpublished Doctor of Philosophy, Victoria University.

NEWMAN, D. G. 1994. Effects of a mouse, *Mus musculus*, eradication programme and habitat change on lizard populations of Mana Island, New Zealand, with special reference to McGregor's skink, Cyclodina macgregori New Zealand Journal of Zoology 21:14.

O'CONNOR, C. E., AND L. H. BOOTH. 2001. Palatability of rodent baits to wild house mice. Department of Conservation, Wellingon, New Zealand.

O'CONNOR, C. E., G. MORISS, AND E. C. MURPHY. 2005. Toxic bait avoidance by mice. Pages 102-105. Proceedings of the 13th Australasian Vertebrate Pest Conference. Wellington, New Zealand.

ROWE-ROWE, D. T., B. GREEN, AND J. E. CRAFFORD. 1989. Estimated impact of feral house mice on Sub-Antactic invertebrates at Marion Island. Polar Biology 9:457-460.

RUSCOE, W. A., AND E. C. MURPHY. 2005. House Mouse. Pages 204-221 *in* C. M. King, editor. The

Handbook of New Zealand Mammals. Oxford University Press, Auckland, New Zealand.

STEPHENSON, B. M., E. O. MINOT, AND D. P. ARMSTRONG. 1999. Fate of moreporks (*Ninox novaeseelandiae*) during a pest control operation on Mokoia Island, Lake Rotorua, North Island, New Zealand. New Zealand Journal of Ecology 23:8.

TERSHY, B. R., C. J. DONLAN, B. S. KEITT, D. A. CROLL, J. A. SANCHEZ, B. WOOD, A. HERMOSILLO, G. R. HOWALD, AND N. BIAVASCHI. 2002. Island conservation in north-west Mexico: a conservation model integrating research, education and exotic mammal eradication. Pages 293-300 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K: IUCN SSC Invasive Species Specialist Group.

THOMAS, B. W., AND R. H. TAYLOR. 2002. A history of ground-based rodent eradication techniques developed in New Zealand, 1959-1993. Pages 301-310 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

TORR, N. 2002. Eradication of rabbits and mice from subantarctic Enderby and Rose Islands. Pages 319-328 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

TOWNS, D. R., AND K. G. BROOME. 2003. From small Maria to massive Campbell: forty years of rat eradications from New Zealand islands. New Zealand Journal of Zoology 30:22.

VAN VUUREN, B. J., AND S. L. CHOWN. 2007. Genetic evidence confirms the origin of the house mouse on sub-Antactic Marion Island. Polar Biology 30:327-332.

VEITCH, C. R. 2002. Eradication of Norway rats (*Rattus norvegicus*) and house mouse (*Mus musculus*) from Motuihe Island, New Zealand. Pages 353-356 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

WANLESS, R. M., A. ANGEL, R. J. CUTHBERT, G. M. HILTON, AND P. G. RYAN. 2007. Can predation by invasive mice drive seabird extinctions? Biology Letters 3:241-244.

WILSON, D. J., E. F. WRIGHT, C. D. CANHAM , AND W. A. RUSCOE. 2007. Neighbourhood analyses of tree seed predation by introduced rodents in a New Zealand temperate rainforest. Ecography 30:105-119.

WITMER, G. W., F. BOYD, AND Z. HILLIS-STARR. 2007. The successful eradication of introduced roof rats (*Rattus rattus*) from Buck Island using diphacinone, followed by an irruption of house mice (*Mus musculus*). Wildlife Research 34:108-115.

**Appendix.** Eradication of house mice from islands worldwide (Updated 2007; the authors would be grateful to be made aware of any omissions or errors in this compilation.)

**Part 1.** Data on operations which have resulted in the eradication of house mice from islands around the world. The methods listed are: A=Aerial, B=Bait stations, H=Hand broadcast, T=Trapping. Toxicants listed are: BM=Brodifacoum, BE=Bromadiolone, DE=Diphacinone, FN=Flocoumafen, PE=Pindone, WN=Warfarin. Countries listed are: AUS=Australia, FRA=France, ICE=Iceland, MAU=Mauritius, NZL=New Zealand, POR=Portugal, ROS=Republic of Seychelles, UK=United Kingdom, US=United States. * = date confirmed after a 2 year confirmation process, # = Method not confirmed, assumed to be bait stations.

| Island | Area (ha) | Country | Started | Methods | Toxicant | Completed | Reference |
|---|---|---|---|---|---|---|---|
| Beacon | 1.2 | AUS | 1997 | B | PE, BM | 1997 | Burbidge and Morris 2002 |
| Bridled | 22 | AUS | 1997 | B | PE, BM | 1997 | Burbidge and Morris 2002 |
| Varanus | 80 | AUS | 1997 | B | PE, BM | 1997 | Burbidge and Morris 2002 |
| Surprise Island | 24 | FRA | 2001 | H | BE | 2006 | F. Courchamp, pers. comm. |
| Flatey Island | 50 | ICE | 1971 | B# | WN | 1971 | Moors et al. 1992 |
| Flat Island | 253 | MAU | 1998 | B | BM | 1998 | Bell 2002 |
| Ile aux Sables | 8 | MAU | 1995 | B, H | BM | 1995 | Bell 2002 |
| Ile Cocos | 15 | MAU | 1995 | B, H | BM | 1995 | Bell 2002 |
| Rasa Island | 60 | MEX | 1994 | A, T | BM | 1994 | Tershy et al. 2002 |
| Allports (Marlborough) | 16 | NZL | 1989 | B | FN | 1991* | Brown 1993 |
| Blumine Marlborough) | 377 | NZL | 2005 | A | BN | 2007* | M. Aviss pers. Comm.. |
| Browns (Hauraki Gulf) | 58 | NZL | 1995 | A | BE | 1997* | Veitch 2002a |
| Enderby (Auckland) | 710 | NZL | 1993 | A | BM | 1995* | Torr 2002 |
| Mana | 217 | NZL | 1989 | A, B | BM, FN | 1991* | Hook and Todd 1992 |
| Mokoia (Lake Rotorua) | 135 | NZL | 2001 | A, H | BM | 2003* | Armstrong et al. 2001 |
| Motuihe (Hauraki Gulf) | 179 | NZL | 1997 | A | BM | 1999* | Veitch 2002b |
| Moturemu (Kaipara) | 5 | NZL | 1992 | B | BM | 1994* | I. McFadden pers. comm. |
| Motutapere (West Coromandel) | 45 | NZL | 1994 | A, B | BM | 1996* | P. Thomson pers. comm. |
| Motutapu (Marlborough) | 2 | NZL | 1989 | B | FN | 1991* | Brown 1993 |
| Mou Waho (Lake Wanaka) | 140 | NZL | 1995 | A, T | BM | 1997* | McKinlay 1999 |
| Ohinau (East Coromandel) | 43 | NZL | 2005 | A | BM | 2006 | J. Roxburgh pers. comm. |
| Papakohatu (Crusoe) | 0.7 | NZL | 1996 | B, T | BM | 1997 | Lee 1999 |
| Pickersgill (Marlborough) | 103 | NZL | 2005 | A | BM | 2007* | M. Aviss pers. comm. |

| Island | Area (ha) | Country | Started | Methods | Toxicant | Completed | Reference |
|---|---|---|---|---|---|---|---|
| Rimariki | 22 | NZL | 1989 | B | BM | 1991 | Veitch & Bell 1990 |
| Whenuakura (Whangamata) | 2 | NZL | 1983 | B | BE | 1984 | Newman 1985 |
| Selvagem Grande | 200 | POR | 2002 | B | BM, HS | 2003 | Oliviera et al. 2003 |
| Frégate | 219 | ROS | 2000 | A | BM | 2002 | Merton et al. 2002 |
| White Cay, Exumas-Bahamas | 15 | UK | 1998 | B | BM | 1998 | Hayes et al. 2004 |

**Part 2.** Data on operations which have not resulted in the eradication of house mice from an island. These operations are listed as: "incomplete" where the work is continuing or confirmation of the eradication has not been obtained; "stopped" where the work was stopped due to a management decision before the planned work was completed; "unsuccessful" where the planned programme was completed and eradication was not successful. The methods listed are: A=Aerial, B=Bait stations, H=Hand broadcast, T=Trapping. Toxicants listed are: BM=Brodifacoum, BE=Bromadiolone, DE=Diphacinone, FN=Flocoumafen, PE=Pindone, WN=Warfarin. Countries listed are: AUS=Australia, FRA=France, ICE=Iceland, MAU=Mauritius, NZL=New Zealand, POR=Portugal, ROS=Republic of Seychelles, UK=United Kingdom, US=United States.

| Island | Area (ha) | Country | Started | Methods | Toxicant | Reference |
|---|---|---|---|---|---|---|
| **INCOMPLETE** | | | | | | |
| Ile Chateau | 250 | FRA | 2002 | A | BM | M. Pascal pers. comm |
| Adele | 87 | NZL | 2007 | A | BM | C. Golding pers. comm. |
| Fisherman | 4 | NZL | 2007 | A | BM | C. Golding pers. comm. |
| Pomona (Lake Manapouri) | 262 | NZL | 2007 | A | BM | ? |
| Rona (Lake Manapouri) | 60 | NZL | 2007 | A | BM | ? |
| Tonga | 8 | NZL | 2007 | A | BM | C. Golding pers. comm. |
| | | | | | | |
| **STOPPED** | | | | | | |
| Silver (Lake Hawea) | 25 | NZL | 1997 | B | BM | S. Thorne pers. comm. |
| Stevensons (Lake Wanaka) | 65 | NZL | 1997 | B | BM | S. Thorne pers. comm. |
| | | | | | | |
| **UNSUCCESSFUL** | | | | | | |
| Varanus | 80 | AUS | 1993 | B | 1080 | Burbidge and Morris 2002 |
| Fajou | 120 | FRA | 2001 | H | BE | M. Pascal pers. comm. |
| St. Paul | 800 | FRA | 1997 | A | BM | Micol and Jouventin 2002 |
| Tromelin | 100 | FRA | 2005 | B, H | BM | ? |
| Patiti (Banded) | 12.8 | NZL | 2004 | B | BM | Bancroft 2004 |

| Island | Area (ha) | Country | Started | Methods | Toxicant | Reference |
|---|---|---|---|---|---|---|
| Haulashore | 6 | NZL | 1991 | B | BM | Thomas and Taylor 2002 |
| Hauturu | 10 | NZL | 1993 | B, H | BM | Glassey 2006 |
| Hokianga (Ohiwa) | 8 | NZL | 2006 | B | PE | D. Paine pers. comm. |
| Limestone (Matakohe) | 37 | NZL | 1996 | A | BM | Ritchie 2000 |
| Limestone (Matakohe) | 37 | NZL | 1997 | A | BM | Ritchie 2000 |
| Limestone (Matakohe) | 37 | NZL | 1998 | A | BM | Brackenbury 2001 |
| Limestone (Matakohe) | 37 | NZL | 1999 | B | BM | P. and C. Mitchell pers. comm. |
| Mokoia | 133 | NZL | 1989 | B | BM | Owen 1998 |
| Mokoia | 133 | NZL | 1996 | A | BM | Dumbell 1998 |
| Quail | 81 | NZL | 2002 | B, H | BM | Bowie 2002 |
| Te Haupa | 6 | NZL | 1993 | B | FN | T. Wilson pers. comm. |
| Bird Island | 101 | ROS | 1996 | B, H | BM | Merton et al. 2002 |
| Curieuse Island | 286 | ROS | 1996 | A | BM | Merton et al. 2002 |
| Denis | 143 | ROS | 2000 | A | BM | Merton et al. 2002 |

## APPENDIX LITERATURE CITED


ARMSTRONG, D. P., J. K. PERROTT, AND I. CASTRO. 2001. Estimating impacts of poison operations using mark-recapture analysis: hihi (*Notiomystis cincta*) on Moikoia Island. New Zealand Journal of Ecology 25:49-54.

BANCROFT, B. 2004. Patiti (Banded) Island rodent eradication 2004. Rotorua, New Zealand: Waimangu Volcanic Valley Ltd and the Department of Conservation.

BELL, B. D. 2002. The eradication of alien mammals from five offshore islands, Mauritius, Indian Ocean. *In* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

BOWIE, M. 2002. Assessment of environmental effects for rat and mouse control on Otamahua/Quail Island, Canterbury: Department of Conservation, New Zealand.

BRACKENBURY, G. 2001. Limestone - Matakohe Island. Southern Bird 7:8-9.

BROWN, D. 1993. Eradication of mice from Allports and Motutapu Islands. Ecological Management 1:19-30.

BURBIDGE, A. A., AND K. D. MORRIS. 2002. Introduced mammal eradications for nature conservation on Western Australian Islands: a review. Pages 64-70 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

DUMBELL, G. 1998. Mokoia Island ecological management strategy: Applied Ecology, Auckland, New Zealand.

GLASSEY, M. 2006. An ecological and historical survey of Hauturu Island, Whangamata: Auckland University of Technology/Bay of Plenty Polytechnic.

HAYES, W. K., R. L. CARTER, S. CYRIL, AND B. THORNTON. 2004. Conservation of an endangered Bahamian rock iguana, 1: population assessments, habitat restoration, and behavioral ecology. Pages 232-257 *in* A. C. Alberts, R. L. Carter, W. K. Hayes, and E. P. Martins, editors. Iguanas: biology and conservation. University of California Press, Berkeley, California USA.

HOOK, T., AND P. TODD. 1992. Mouse eradication on Mana Island. Page 33 *in* D. Veitch, M. Fitzgerald, J. Innes, and E. Murphy, editors. Proceedings of the national predator management workshop. Department of Conservation, Wellington, New Zealand.

LEE, M. 1999. Biota of seven islets off Waiheke Island, Inner Hauraki Gulf. Tane 37:99-136.

MCKINLAY, B. 1999. Eradication of mice from Mou Waho, Lake Wanaka. Ecological Management 7:1-5.

MERTON, D., G. CLIMO, V. LABOUDALLON, S. ROBERT, AND C. MANDER. 2002. Alien mammal eradication and quarantine on inhabited islands in the Seychelles. *In* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

MICOL, T., AND P. JOUVENTIN. 2002. Eradication of rats and rabbits from Saint-Paul Island, French Southern Territories. Pages 199-205 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. Gland, Switzerland: IUCN.

MOORS, P. J., I. A. E. ATKINSON, AND G. H. SHERLEY. 1992. Reducing the rat threat to island birds. Bird Conservation International 2:93-114.

NEWMAN, D. G. 1985. The apparent loss of the Whenuakura Island tuatara population, Whangamata Islands wildlife sanctuary. Wellington: New Zealand Wildlife Service.

OLIVEIRA, P., R. TROUT, D. MENEZES, AND P. GERALDES. 2003. Recuperaçâo dos habitates terrestres da Selvagem Grande. Pages 53-60 *in* J. L. Rodriguez-Luengo, editor. Control de Vertebrados Invasores en Islas de España y Portugal. Santa Cruz de Tenerife, Spain.: Consejería de Medio Ambiente y Ordenación Territorial del Gobierno de Canarias.

OWEN, K. 1998. Removal and reintroduction of North Island weka (*Gallirallus australis greyii*) to Mokoia Island as a result of a Talon 7/20 cereal-based aerial poison drop. Ecological Management 6:8.

RITCHIE, J. 2000. Matakohe - Limestone Island Scenic Reserve restoration plan: natural logic evironmental managament ltd., Auckland, New Zealand.

TERSHY, B. R., C. J. DONLAN, B. S. KEITT, D. A. CROLL, J. A. SANCHEZ, B. WOOD, A. HERMOSILLO, G. R. HOWALD, AND N. BIAVASCHI. 2002. Island conservation in north-west Mexico: a conservation model integrating research, education and exotic mammal eradication. Pages 293-300 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K: IUCN SSC Invasive Species Specialist Group.

THOMAS, B. W., AND R. H. TAYLOR. 2002. A history of ground-based rodent eradication techniques developed in New Zealand, 1959-1993. Pages 301-310 *in* C. R. Veitch and M. N. Clout, editors. Turning the Tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

TORR, N. 2002. Eradication of rabbits and mice from subantarctic Enderby and Rose Islands. Pages 319-328 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

VEITCH, C. R. 2002a. Eradication of Norway rats (*Rattus norvegicus*) and house mouse (*Mus musculus*) from Browns Island (Motukorea), Hauraki Gulf, New Zealand. Pages 350-352 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

VEITCH, C. R. 2002b. Eradication of Norway rats (*Rattus norvegicus*) and house mouse (*Mus musculus*) from Motuihe Island, New Zealand. Pages 253-356 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN, Gland, Switzerland and Cambridge, U.K.: IUCN SSC Invasive Species Specialist Group.

VEITCH, C. R., AND B. D. BELL. 1990. Eradication of introduced animals from the islands of New Zealand. *In* D. R. Towns, C. H. Daugherty, and I. A. E. Atkinson, editors. Ecological restoration of New Zealand islands. Department of Conservation, Wellington, New Zealand.

# THE PATH TO ERADICATION OF THE GAMBIAN GIANT POUCHED RAT IN FLORIDA

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***Abstract:*** A thriving population of Gambian giant pouched rats became established on Grassy Key, a 550-ha island in Florida, following escape(s) from an exotic pet breeder. After existence of the population was verified, computer models indicated that Gambian giant pouched rats could successfully invade a large portion of North America if they reached the mainland. This largest of rat species is highly prolific, and its dispersal to the mainland could result in substantial negative impacts to agriculture, environment, and wildlife. Additionally, Gambian giant pouched rats are known vectors of a variety of diseases transmissible to humans and livestock. The first action to counter the severe and immediate threat of dispersal was to rapidly develop the information necessary on which to base an eradication program. The information included detection and monitoring technologies, population indexing methodologies, population distribution, habitat preferences, trapping methodology, acceptance of bait matrices, and efficacy tests of toxicants, and bait stations that minimize exposure to native species. With these tools forming a foundation, a pilot eradication was funded for Crawl Key, a 150-ha key adjoining Grassy Key to which the species expanded its range. The aims of the pilot eradication were to test and fine-tune the methods prior to implementing full-scale eradication on Grassy Key. No Gambian giant pouched rats were found in two subsequent surveys of Crawl Key. Further surveys of Grassy Key were used to refine bait station densities for the full scale eradication effort implemented on Grassy Key in spring 2007. The eradication effort is on-going.

***Key Words:*** bait station, *Cricetomys gambianus*, eradication, Florida Keys, Gambian giant pouched rat, invasive species, monitoring, rodenticide.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Breeding populations of non-native vertebrate species are regularly identified in Florida. In fact, Florida, along with Hawaii, have the most severe invasive species problems in the United States (US Congress 1993). The negative impacts inflicted by exotic species on native species and ecosystems might be exceeded only by human-caused habitat destruction (Wilcove et al. 1998, Parker et al. 1999). The impacts from many introduced species are unknown or not readily perceived by the public. The realization of an introduced species' existence, and perhaps even its potential for severe impacts, may not occur until after the species has been established for some time. Such was the case with the Gambian giant pouched rat (*Cricetomys gambianus*) in Florida.

Eight Gambian giant pouched rats apparently escaped from an exotic pet breeder on Grassy Key, Florida around 1999 (Perry et al. 2006). A local resident brought the Florida population to the attention of authorities after media reports had associated Gambian giant pouched rats in the US pet trade with an outbreak of monkeypox (Centers for Disease Control 2003). Unconfirmed sightings suggested possible dispersal to other Keys.

The Gambian giant pouched rat had the potential to become a highly destructive exotic species in the United States (US), particularly in agriculture. The species is among the largest members of the rat family *Muridae*, with males achieving weights as high as 2.8 kg (Rosevear 1969). They are omnivorous, consuming a variety of vegetables and fruits, insects, crabs, and snails

(Ajayi 1975, Smithers 1983, Fiedler 1988, Fiedler 1994). Gambian giant pouched rats are also highly fecund, with gestation times ranging from 27-42 days and 4-5 litters/year of 1-5 offspring (Rosevear 1969, Ajayi 1975, Hayssen et al. 1993). In an African agricultural setting, 42 Gambian giant pouched rats were removed from a 0.2 ha field of young peas in Zimbabwe (Smithers 1983, Fiedler 1994). Besides monkeypox, members of this genus have been linked to potentially pathogenic zoonoses such as leptospirosis, bartonellosis, murine typhus, Q-fever and trypanasomiasis (Gretillat et al. 1981, Fiedler 1988, Hutin et al. 2001, Herder et al. 2002, Machang'u et al, 2004),

The Gambian rats large size, high fecundity, omnivorous diet, and potential as vectors of serious diseases made this species an immediate threat to the indigenous ecological communities and human interests within the Florida Keys. Moreover, ecological modeling demonstrated that Gambian giant pouched rats could invade and establish viable populations throughout Peninsular Florida, the US Gulf Coast, and beyond to Central America (Peterson et. al. 2006). Wider areas of North America were also shown to be vulnerable to Gambian giant pouched rat population establishment at lesser probabilities.

Recognition of an invasive species as a severe threat to natural ecosystems, agriculture, and human health and safety does not ensure that an eradication program will take place, nor that such a program once implemented would be successful. A successful eradication effort requires development of pertinent information, appropriate techniques, and the political will (including funding) to proceed effectively. Here, we describe this path culminating in an eradication effort for Gambian giant pouched rats in Florida (US), and the status of that effort as of September 2007.

## IN THE BEGINNING

Until the monkeypox outbreak, Gambian giant pouched rats were popular animals in the pet trade. Like many other species, they were captive-bred in Florida (many of these other species have also successfully established breeding populations in Florida). As already indicated, escapees from a pet breeder on Grassy Key formed the foundation for a burgeoning population. Without notification of their existence, one can only speculate how long the population would have existed, grown, and spread before action was taken, and whether that action would have been in time to prevent their dispersal to the mainland.

The initial step towards eradicating Gambian giant pouched rats was to confirm the existence of a breeding population, which was carried out by a graduate student from Texas A&M University already in the area researching Lower Keys marsh rabbits (*Sylvilagus palustris hefneri*) (Perry et al. 2006). The population's existence prompted the already mentioned computer modeling that projected wide dispersal on the continent if the species reached the mainland (Peterson et al. 2006).

## GATHERING INFORMATION, DEVELOPING TECHNOLOGIES AND STRATEGIES

Once a Gambian giant pouched rat population was confirmed as established, the next step towards addressing the Gambian giant pouched rat population was to generate information in areas essential for commencing an eradication program: (1) methods to detect and monitor populations, (2) distribution, (3) habitat preferences, (4) development of control methods, and (5) influence of non-target species on detection and management.

### Detecting and Monitoring Gambian Giant Pouched Rats

Having a rapid means to detect and survey Gambian giant pouched rats was essential for monitoring distribution and relative abundance. Several existing methods were tested for their ability to detect the presence of Gambian giant pouched rats, including live trapping, remote digital cameras and tracking tiles. Cameras and tracking tiles served well to detect Gambian giant pouched rats while also accommodating large numbers of nontarget species. Simultaneous monitoring of non-target species was an important consideration, as this would provide valuable information for assessing and reducing non-target hazards from control methods, and for reducing non-target interference with control technologies. Both cameras and tracking tiles were also suitable for field application, and produced data appropriate for use in a general indexing paradigm whereby populations could be monitored in a statistically valid fashion (Engeman 2005).

Although the track plate materials and methodology were developed to most efficiently record tracks with a minimum of resources, the method was still substantially more labor-intensive than using remote digital cameras. Therefore, track

plates were considered for operational application only in situations where the risk of theft or vandalism was too great for deploying cameras (Engeman et al. 2006). Motion-triggered digital cameras served particularly well to detect Gambian giant pouched rats while providing data suitable for monitoring abundance. Digital cameras could record a large number of visits, with batteries usually lasting 3-4 days. Their ability to record Gambian giant pouched rats was not hindered by non-target species saturation at camera stations. Cameras also provided a reliable tool to evaluate control efficacy, detect varying Gambian giant pouched rat abundances throughout Grassy Key, and optimize of timing and placement of control devices in a fashion similar to the strategies that greatly improved control efficacy and efficiency for managing predators of sea turtle nests (Engeman et al. 2003, Engeman et al. 2005). Camera surveys also verified Gambian giant pouched rat survival on Grassy Key in the wake of highly destructive Hurricane Wilma. Although much of Grassy Key was inundated by more than 1 m of salt water from the storm surge, the continued presence of Gambian giant pouched rats was readily demonstrated.

Trapping was applied to address multiple needs: (1) detect the presence of Gambian giant pouched rats, (2) refine the information on their distribution, (3) refine trapping techniques including bait and trap placement, and (4) evaluate its impacts to and from non-target species. Gambian giant pouched rats were relatively easily captured using raccoon-sized cage traps baited with peanut butter and/or fruit. However, relative to the numbers of Gambian giant pouched rats captured, large numbers of non-target animals were captured, primarily raccoons (*Procyon lotor*), but also Virginia opossums (*Didelphis virginiana*), black rats (*Rattus rattus*), and the occasional feral cat (*Felis silvestris catus*). About five and a half times as many non-target animals were captured as Gambian giant pouched rats. Thus, non-target trap saturation impaired Gambian giant pouched rat trapping efficiency (Engeman et al. 2006). Also, Gambian giant pouched rats were not captured in traps that had previously captured a raccoon without a thorough cleansing, which could be an important consideration for future trapping efforts, but previous capture of a Gambian giant pouched rat did not appear to affect results (Engeman et al. 2006). A similar effect was documented in Hawaii where black rats avoided traps that had previously captured a mongoose (Tobin et al. 1994).

**Distribution and Habitat Preferences**

Multiple detection and survey methods were applied to identify the Gambian giant pouched rat distribution among the keys. Islands with unconfirmed reports of sightings were a priority for surveillance. Many of the unconfirmed sightings occurred on neighboring Keys. Although originally separated by saltwater, a series of earth-filled causeways connects Grassy Key to its neighbors to the west, whereas an 11-km bridge connects it to neighboring Keys to the east. Thus, keys nearby Grassy Key were also the focus of surveys, as were the islands involved in the transfer of trash and hurricane debris from Grassy Key. The short distances to neighboring keys, especially those connected to Grassy Key by the earth-filled causeways provide access for Gambian giant pouched rats to move to new islands, whereas refuse hauling could offer ready, but inadvertent, human-aided transport to other keys and mainland. Thus, the vicinity of the transfer station on Long Key (11 km east of Grassy Key) where Grassy Key trash collection is transported was surveyed by multiple methods, as was the site on Knights Key (5 km west of Grassy Key) where debris from Grassy Key following Hurricane Wilma was piled for later transfer (Engeman et al. 2006).

Outside of Grassy Key, Gambian giant pouched rats were found only on Crawl Key, the first key west of Grassy Key, and thereby demonstrating that the soil-filled causeways had been used for dispersal to new locations (Engeman et al. 2006). Fortunately no Gambian giant pouched rats were found on keys connected to Crawl Key by earth-filled causeways, nor on Long or Knights Keys with transfer sites for refuse or hurricane debris.

On Grassy and Crawl Keys, Gambian giant pouched rats were observed only in drier rockland hammock habitat (Florida Natural Areas Inventory 1990). Gambian giant pouched rats were found in a 2-km band of this habitat on Grassy Key extending east and west from the site where the escape originated. However, they were not observed in similar, apparently suitable habitats towards the eastern end of Grassy Key. The preferred habitat for Gambian giant pouched rats coincided with the habitat most suitable for human development. While Gambian giant pouched rats were found in undeveloped rockland hammock habitat, human development of this habitat greatly enhanced these areas for Gambian giant pouched rat occupation through provision of a greater bounty of resources, such as refuse, pet food, fresh water, and refugia. They did not appear to inhabit wet shrub and

mangrove habitats. A small radio-telemetry data set also confirmed the observations by the various detection methods concerning habitat preferences. Moreover, radio-telemetry demonstrated Gambian giant pouched rats, males in particular, are capable of overnight movements of nearly 1 km (Engeman et al. 2006). However, information from their native African range suggests that the Gambian giant pouched rats do not move very far; usually only repeated movements between a food source and their burrow (Smithers 1983).

**Bait and Toxicant Development**

Live-trapped Gambian giant pouched rats were used to rapidly evaluate bait acceptance and efficacy. The toxicants tested included 2.0% zinc phosphide (ZP), an acute toxicant, in a peanut butter and horse sweet mix matrix (corn, oats, molasses) and four commercial anticoagulant baits: Ramik® mini-bars (0.005% diphacinone, first generation anticoagulant), Contrac® (0.005% bromadiolone, second generation anticoagulant), Havoc® (0.005% brodifacoum, second generation anticoagulant), and d-Con® (0.0025% difethialone, second generation anticoagulant). Among the commercial anticoagulant baits, the Gambian giant pouched rats showed the greatest acceptance for the Ramik® mini-bars over the three second generation anticoagulants baits. All rats consumed this bait and died in 5-11 days. All rats presented with the ZP bait died in $\leq$43 h (most in < 24 h), after consuming a small amount (mean = 7.3 g) of the ZP bait (Engeman et al. 2006).

Based on those results, two rodenticide baits appeared most valuable for use on Gambian giant pouched rats: the first generation anticoagulant, 0.005% diphacinone (Ramik mini-bars), and the acute 2% ZP bait (mixed with peanut butter and horse sweet mix). The Ramik bait would need to be presented continuously for at least a two-week period to assure consumption of a lethal dose, whereas, a small amount of the acute ZP bait consumed in a single feeding would be lethal.

A commercial bait station (Protecta®, Bell Laboratories Inc., Madison, Wisconsin) and a custom-designed bait station made of PVC pipe (Engeman et al. 2006) were tested using untreated bait to determine bait accessibility for Gambian giant pouched rats, and exclusion of native species. Over 600 photographic observations of the PVC pipe bait station demonstrated delivery of bait to Gambian giant pouched rats while minimizing exposure to native species. The commercial bait station appeared to exclude native mammalian species, but also restricted adult Gambian giant pouched rats from access (Engeman et al. 2006).

**Influence of Non-Target Species**

Sympatric populations of non-target species, especially raccoons, existed in great abundance, presenting a threat to improperly configured control and monitoring devices intended for Gambian giant pouched rats. Control devices must exclude non-target species, and monitoring devices needed to accommodate many non-target visits and still be able to detect Gambian giant pouched rats.

The camera methodology used to detect Gambian giant pouched rats was not hindered by large numbers of non-target animals. On the other hand, based on camera observations, non-target species could potentially remove all bait from bait stations (and probably die) before Gambian giant pouched rats would have a chance at the bait. However, this problem was averted by the innovative bait station design. The same could not be said for trapping. Trapping would still be required in areas where bait was not permitted by property owners, where rats consumed a sublethal dose and became aversively conditioned to the bait, or for rats not attracted to the bait stations to begin with. In these situations, non-target animals could pose significant obstacles to Gambian giant pouched rat removal through trapping.

## PILOT ERADICATION ON CRAWL KEY

Prior to implementing full-scale eradication on Grassy Key, a pilot eradication project was implemented to test and fine-tune the methods on Crawl Key adjoining Grassy Key where the species expanded its range. In addition to the governmental entities involved in the project, partial funding also was provided by the Wildlife Foundation of Florida, Inc. The information accumulated on Gambian giant pouched rats and control methods was formulated into an eradication plan and tested on this small Key.

Because the ZP bait was readily accepted with only minimal consumption required for a lethal dose (thereby increasing the likelihood a lethal dose would be consumed at the bait station with a single visit), it was selected as the bait of choice. A 40 x 40 m grid of bait stations was established for Crawl Key, and was based on the radio-telemetry data, information from the literature, and personal experience (GW) in eradication of other rat species from other islands. Pre-baiting using the same bait matrix without the toxicant is common and often

recommended on the labels for ZP baits to help prevent "bait-shyness" whereby animals become somewhat sick from a sublethal dose, and decline to eat that bait again (for a review, see Salmon et al. 2000). Therefore, pre-baiting in the PVC bait stations was done for three days using the bait matrix without the toxicant. Following that, any remaining bait matrix was removed and toxic bait (2% ZP) was applied for seven days. All of Crawl Key was baited at the same time (97 bait stations).

Preliminary camera surveys following the completion of the pilot eradication found no evidence of Gambian giant pouched rats remaining on Crawl Key. However, the highly destructive Hurricane Wilma struck in the intervening time. Its storm surge over washed much of the island with up to 1 m of water, which may have also contributed to their mortality. Further monitoring of Crawl Key is required, however, to assure the successful eradication from that island.

## ERADICATION ON GRASSY KEY

### Planning the Eradication

The criteria for a successful eradication (see Parkes and Murphy 2003, Engeman et al. 2006) were considered obtainable and an eradication effort subsequently was commenced on larger Grassy Key, the location of the primary Gambian giant pouched rat population. Surveys on Grassy Key following Hurricane Wilma verified the survival of the Gambian giant pouched rat population with, possibly, a greater occupied area. That the rats survived the hurricane on Grassy Key provided sound evidence that they probably could have survived also on Crawl Key, and their absence there was most likely due to bait consumption or because a longer period of monitoring is required.

In late 2006 and early 2007, the population surveys on Grassy Key were completed and used to design bait station density and distribution across the island. Not all of Grassy Key appeared to hold Gambian giant pouched rats, especially areas with inferior (very wet) habitat. Nevertheless, baiting was conducted throughout the entirety of the island where high water was not an issue. However, bait station density was varied between two levels according to the probability that Gambian giant pouched rats were present in the vicinity. The areas known to support Gambian giant pouched rats had a bait station placement design on a 40 x 40 m grid, whereas the perceived marginal areas for Gambian giant pouched rats had a 50 x 50 m grid. A public meeting, mailings, and door-to-door visits were conducted to get public approval among the 494 property owners. A toll-free hotline was established to provide information on eradication time lines and progress. Still, some property owners either would not grant permission to enter their property or would not allow toxic bait on their property. Baiting was planned to be conducted as close to those properties as legally permissible. Even so, trapping was thus anticipated to be a necessary addition for eradication.

### Implementing the Eradication

To avoid hazards to, and interference from non-target species during baiting, live-trapping was conducted to safely remove non-target animals to a nearby island. The South Florida Water Management District funded private contractors to cut the necessary trails for establishing the bait station grids. Next, 1,000 bait stations were installed at the predetermined spacings, a process that extended from January through May due to personnel constraints and competing priorities. Because of the large number of bait stations, all bait stations on the entire island could not be serviced simultaneously. Baiting with the 2% ZP formulation was conducted in May-June 2007 in a "rolling front" strategy. The island was divided longitudinally into zones. Bait was applied to one zone at a time, moving from east to west. Pre-baiting was again applied for three days using the nontoxic bait matrix, followed again by seven days of baiting with toxic bait.

### Current Status

In areas holding Gambian giant pouched rats, it appeared that high percentages of non-toxic bait were commonly removed from bait stations during pre-baiting, but much lower quantities of toxic bait were removed, probably indicating single feeding mortality or a relatively low acceptance rate. It became quickly clear that some Gambian giant pouched rats remained after the baiting effort. Camera surveys following completion of baiting revealed at least four hotspots of Gambian giant pouched rat activity. These sites have been primarily associated with residential areas where numerous alternative food sources, such as pet food, are available. A few activity sites also have been associated with properties where the owners did not permit the use of toxicants. These final hotspots are being addressed with intensive trapping using cantaloupe as a food bait in the cage traps. The surrounding bait stations are also being re-baited with the ZP bait, mixing the ZP with

horse sweet mix and cantaloupe oil. This approach will hopefully be successful, especially as the translocation of raccoons and opossums away from Grassy Key has reduced non-target animal interference with traps to much lower levels. One hotspot, where the property owner did not allow toxicants and also had fresh water and pet food available, has been repeatedly trapped, but Gambian giant pouched rats remain in the area. Because eradication is proving very difficult under the conditions on Grassy Key, alternative rodenticides are being investigated. However, preliminary trials with Ramik mini-bars and Ramik Green pellets (diphacinone) and captive rats being fed a diverse diet has shown this rodenticide bait to be inadequately palatable to the Gambian giant pouched rats. A brodifacoum pelleted bait (Brodifacoum 25–Conservation) is also being investigated, and preliminary results are promising. If an effective, alternative bait is found, the entire island will be baited using the bait station grid once approval is obtained from the Florida Department of Agriculture.

## DISCUSSION

Gambian giant pouched rats will be considered eradicated when intensive surveys do not reveal the presence of rats for two years, with the caveat that vigilance and periodic surveys will continue past that time to ensure no remnants of the population have gone undetected and survived to breed. The areas of any detected rats would be targeted with intensive control efforts. If this invasive species can be successfully eradicated from the Florida Keys, hopefully, this would help reduce the general reluctance of managers to attempt eradications of other invasive species in Florida (see, for example the comments by Donlan et al. 2003).

### Teamwork, Resources, and the Future

The logic and flow described here for this eradication effort may make it seem as though the path to Gambian giant pouched rat eradication was a smooth continuum once the problem was identified. In reality, it was a series of fits and starts, beginning in August 2004 and continuing to date. There was no sizable and continuous block of funding available to develop the necessary information and implement an eradication effort. Funding and in-kind resources were provided from several federal, state, and local government entities, as well as private concerns. Work towards eradication has been conducted to the fluctuating level of the resources available.

Work will continue towards the eradication of Gambian giant pouched rats on Grassy Key as long as resources are available. Once the eradication effort appears to have been successful, however, that does not imply the eradication effort would be complete. At least two years of monitoring for Gambian giant pouched rats should be applied to both Grassy and Crawl Keys. Moreover, similar monitoring efforts should be applied to transfer sites for refuse from Grassy Key, including the mainland landfill(s). While investigation of unconfirmed, but credible reports of sightings on other keys did not locate any Gambian giant pouched rats, these areas should also receive continued monitoring to help insure no individuals from Grassy Key are surviving elsewhere in the Florida Keys. This ongoing monitoring effort will require funding, as would a rapid response to any confirmed locations of Gambian giant pouched rats. Hopefully, complacency with the accomplishments so far would not undermine availability of necessary resources to continue mop-up work of remaining rats and to do the follow-up monitoring portion of the effort to its conclusion. Lack of continued vigilance could result in the hard work and expended resources to date being undone, or worse, eventual Gambian giant pouched rat dispersal to the mainland.

Although obtaining adequate resources to maintain steady work towards eradication is challenging, the effort will, hopefully, be successful in the end. If successful, the approach developed here could serve as a model for preparing control or eradication efforts for other potentially destructive invasive species. This is especially true in Florida, which often serves as the gateway to North America for non-native introductions (US Congress 1993). Similar to the case of the Gambian giant pouched rat, the rapid development of detection, monitoring, and control methods could quickly lead to implementation of successful control or eradication procedures while a practical opportunity exists to contain and/or remove their populations.

## LITERATURE CITED

AJAYI, S. S. 1975. Observations on the biology, domestication, and reproductive performance of the african giant pouched rat *Cricetomys gambianus* (Waterhouse) in Nigeria. Mammalia 39:344-364.

CENTERS FOR DISEASE CONTROL. 2003. Multistate outbreak of monkeypox—Illinois, Indiana, Kansas, Missouri, Ohio, and Wisconsin, 2003. Morbidity and Mortality Weekly Report 52:642-644.

DONLAN, C. J., B. R. TERSHY, K. CAMPBELL, AND F. CRUZ. 2003. Research for requiems: the need for more collaborative action in invasive species management and conservation. Conservation Biology 17:1850-1851.

ENGEMAN, R. M. 2005. A methodological and analytical paradigm for indexing animal populations applicable to many species and observation methods. Wildlife Research. 32:203-210.

ENGEMAN, R. M., R. E. MARTIN, B. CONSTANTIN, R. NOEL, AND J. WOOLARD. 2003. Monitoring predators to optimize their management for marine turtle nest protection. Biological Conservation 113:171-178.

ENGEMAN, R. M., R. E. MARTIN, H. T. SMITH, J. WOOLARD, C. K. CRADY, S. A. SHWIFF, B. CONSTANTIN, M. STAHL, AND J. GRINER. 2005. Dramatic reduction in predation on sea turtle nests through improved predator monitoring and management. The Oryx 39:318-326.

ENGEMAN, R. M., J. W. WOOLARD, N. D. PERRY, G. WITMER, S. HARDIN, L. BRASHEARS, H. T. SMITH, B. MUIZNIEKS, AND B. U. CONSTANTIN. 2006. Rapid assessment for a new invasive species threat: the case of the Gambian giant pouched rat in Florida. Wildlife Research 33:439-448.

FIEDLER, L. 1988. Rodent problems in Africa. Pages 35-65 *in* I. Prakash, editor. Rodent pest management, CRC Press, Boca Raton, Florida, USA.

FIEDLER, L. 1994. Rodent pest management in eastern Africa. FAO Plant Production and Protection Paper 123. FAO, Rome.

FLORIDA NATURAL AREAS INVENTORY (FNAI). 1990. Guide to the natural communities of Florida. Florida Natural Areas Inventory and Florida Department of Natural Resources, Tallahassee, Florida, USA.

GRETILLAT, S., X. MATTEI, AND B. MARCHAND. 1981. A new rickettsiale of Gambia rats (*Cricetomys gambianus*) in Senegal: *Grahamella kaniae* n. sp. (Bartonellacae), Rev Elev Med Vet Pays Trop. 34:383-389.

HAYSSEN, V., A. VAN TIENHOVEN, AND A. VAN TIENHOVEN. 1993. Asdell's patterns of mammalian reproduction: a compendium of species specific data. Comstock/ Cornell University, Ithaca, NY, USA.

HERDER, S., G. SIMO, S. NKININ, AND F. NJIOKOU. 2002. Identification of trypanosomes in wild animals from southern Cameroon using the polymerase chain reaction (PCR). Parasite 9:345-349.

HUTIN, Y., R. J. WILLIAMS, P. MALFAIT, R. PEBODY, V. N. LOPAREV, S. L. ROPP, M. RODRIGUEZ, J. C. KNIGHT, F. K. TSHIOKO, A. S. KHAN, M. V. SZCZENIOWSKI, AND J. J. ESPOSITO. 2001. Outbreak of human monkeypox, Democratic Republic of Congo, 1996-1997. Emerging Infectious Disease 7:434-438.

MACHANG'U, R. S., G. F. MGODE, J. ASSENGA, G. MHAMPHI, B. WEETJENS, C. COX, R. VERHAGEN, S. SONDIJ, M. G. GORIS, AND R. A. HARTSKEERL. 2004. Serological and molecular characterization of leptospira serovar Kenya from captive African giant pouched rats (*Cricetomys gambianus*) from Morogoro Tanzania. FEMS Immunology and Medical Microbiology 41:117-121.

PARKER, I. M., D. SIMBERLOFF, W. M. LONSDALE, K. GOODELL, M. WONHAM, P. M. KAREIVA, M. H. WILLIAMSON, B. V. HOLLE, P. B. MOYLE, J. E. BYERS, AND L. GOLDWASSER. 1999. Impact: toward a framework for understanding the ecological effects of invaders. Biological Invasions 1:3-19.

PARKES, J., AND E. MURPHY. 2003. Management of introduced mammals in New Zealand. New Zealand Journal of Zoology 30:335-359.

PERRY, N. D., B. HANSON, W. HOBGOOD, R. L. LOPEZ, C. R. OKRASKA, K. KAREM, I. K. DAMON, D. S. CARROLL. 2006. New invasive species in southern Florida: Gambian rat (*Cricetomys gambianus*). Journal of Mammalogy 87:262-264.

PETERSON, A. T., M. PAPES, M. REYNOLDS, N. D. PERRY, B. HANSON, R. REGNERY, C. HUTSON, B. MUIZNIEK, I. DAMON, D. AND CARROLL. 2006. Native range ecology and invasive potential of *Cricetomys* rats in North America. Journal of Mammaolgy 87:427-432.

ROSEVEAR, D. R. 1969. The rodents of West Africa. British Museum of Natural History, London.

SALMON, T., D. A. WHISSON, AND W. P. GORENZEL. 2000. Use of zinc phosphide for California ground squirrel control. Pages 346-357 *in* T. A. Salmon and A. C. Crabb, editors. Nineteenth Vertebrate Pest Conference, University of California Davis, California, USA.

SMITHERS, R. 1983. The mammals of the Southern African subregion. University of Pretoria, Pretoria, South Africa.

TOBIN, M. E., R. M. ENGEMAN, AND R. T. SUGIHARA. 1994. Effects of initial rat captures on subsequent capture success of traps. Pages 101-105 *in* W. S. Halverson and A. C. Crabb, editors. Sixteenth Vertebrate Pest Conference, Santa Clara, California, USA.

US CONGRESS. 1993. Harmful non-indigenous species in the United States. Office of Technology Assessment, OTA-F-565, Government Printing Office, Washington, D.C., USA.

WILCOVE, D. S., D. ROTHSTEIN, J. DUBOW, A. PHILLIPS, AND E. LOSOS. 1998. Assessing the relative importance of habitat destruction, alien species, pollution, over-exploitation, and disease. BioScience 48:607-616.

# RESEARCH AND DEVELOPMENT OF A NEW RODENTICIDE FOR NUTRIA

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***Abstract:*** The nutria (*Myocastor coypus*) is a large semi-aquatic rodent, declared as one of the 100 worst invasive species in the world by the IUCN Invasive Species Specialist Group in 2000. Through USDA Small Business Innovative Research Phase I (Mach 2004) and Phase II (Mach 2006) funding, Genesis is developing a new nutria rodenticide as an alternative to the currently-registered zinc phosphide bait. Our course of research and development is described herein: live-trapping, laboratory non-toxic choice studies, laboratory toxicity bioassays, and field toxicity studies. Two active ingredients and two bait types proved to be successful, however, only cholecalciferol in each of two bait types will be tested in future field studies.

***Key Words:*** cholecalciferol, coypu, invasive species, *Myocastor coypus*, nutria, rodenticide.

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## INTRODUCTION

The nutria or coypu is a large (over 6 kg), semi-aquatic rodent with a voracious appetite and high reproductive potential. It is a rodent native to South America that was introduced to the United States (US) in fur farms; however, they have become established in the wild in numerous states. The marsh habitat of Louisiana proved to be ideal and populations exploded, reaching an estimated 20 million animals in less than 20 years (Evans 1970).

Numerous studies of the wetland environments of Louisiana have documented the deleterious effects of nutria grazing. Nutria herbivory has been documented on species of bald cypress (*Taxodium distichum*) (Conner and Toliver 1987), *Sagittaria latifolia* and *S. platyphylla* (Grace and Ford 1996, Evers et al. 1998), *Spartina patens* and *S. alterniflora* (Taylor et al. 1997, Ford and Grace 1998), and many other species of marsh vegetation (Fuller et al. 1985, Foote and Johnson 1992, Taylor and Grace 1995).

Trapping of nutria for their pelts formed the backbone of the Louisiana trapping industry from the 1960s until the early 1980s when price for fur fell dramatically. Since then, the annual trapping harvest has dwindled to 29,544 in the 2000-2001 season. Reports of damage to wetland habitats emerged in the late 1980s. In November 2002, an incentive-payment program for the nutria control was initiated through funding by a 20-year program, Coastal Wetland Planning, Protection, and Restoration Act (CWPPRA) with the intent of curbing the amount of marsh herbivory. The program has been successful in that an approximate mean of 300,000 nutria tails per year have been turned in for payment. Adjustment has been made in the incentive payment, currently $5 per tail, to entice additional control and retain a high level of trapping pressure.

Through USDA Small Business Innovative Research (SBIR) grants, we are developing a unique buoyant rodenticide to be applied aerially or by hand spot-baiting for nutria control. A unique toxicant and bait form will provide an alternative to current control methods.

## PROJECT METHODS AND RESULTS

Prior to the initiation of any of the studies, several nutria were live-captured by personnel of Genesis Laboratories, Inc. and Louisiana Department of Wildlife and Fisheries, Fur and Refuge Division, New Iberia, Louisiana. All nutria were captured at Salvador Wildlife Management Area, an area managed by the State and located near Paradis, Louisiana, southwest of New Orleans. They were captured by hand or by large fishing net while driving an airboat alongside the swimming or running nutria. As they were captured, they were placed in portable wire mesh cages. The nutria were transported to the Department of Wildlife and Fisheries office in New Iberia or to Genesis Laboratories, Inc., in Wellington, Colorado, for holding and laboratory research and development. IACUC approved protocol numbers N04015 and N07004 were reviewed and followed for appropriate animal care.

Nutria were held individually in stainless steel cages with a floor area of 576 $in^2$ (24 x 24 x 15 in)

or in small groups in larger stock tanks with a floor area of approximately 1,200 $in^2$. Nutria were offered vegetables *ad libitum* as food and a water source. Bedding was changed at least once each week.

The project progressed with the following studies: choice bioassays, toxicant efficacy, and secondary toxicity. Following is a short description of the methods and the results of each study.

**Choice Bioassays**

Native vegetation was intended to be the challenge diet that we would use for all choice tests, but it quickly became evident that the nutria could consume large amounts and transportation acquisition was very difficult. If nutria are eating normally, they can consume as much as 25% of their body weight each day (Christen 1978). We began using raw russet potato as the "challenge diet."

Due to the large amount of food that the nutria could consume, we used limited exposure periods during the work hours of each study day. For example, for a 2-day test, we would offer the challenge diet (potato) and the test substance for approximately 4-6 hours during each study day. We were limiting the exposure period of the test substance to assure that the nutria would not consume all of one food item and then begin eating the less preferred item. Allowing consumption of an entire diet item would give false results on the performance of the test diets. To achieve a valid data set, spilled diets were retrieved from the bedding of each cage. Test substance acceptance (%) was calculated with the following formula:

$$\frac{\text{Total Test Substance Consumption (g)}}{\text{Total Test Substance Consumption (g) + Total Challenge Diet Consumption (g)}} \times 100\%$$

We began testing different formulas, vegetables, fruits, commercially available pet feeds, modified floating fish baits, and many other food-grade formulations. Many of the formulations did not float and were dismissed from additional testing.

Finally, two types of formulas were chosen as possibilities, a vegetable bait (celery or carrot) to be covered with the toxicant, and a gum-based formula in to which the toxicant could be formulated.

**Toxicant Efficacy**

We have evaluated the use of four different toxicants that are currently registered for other rodent species and uses throughout the world. We evaluated bromethalin, diphacinone, cholecalciferol, and sodium monofluoroacetate. Not all of these toxicants are currently registered in the US, but their potential for this particular use was justifiable for research purposes. Zinc phosphide was not considered as it is already registered and used as a topical toxicant on vegetables for nutria control.

Nine nutria were tested with each of the following active ingredients: cholecalciferol, bromethalin, diphacinone, and sodium monofluoroacetate (Table 1). These data provided initial toxicity of these compounds to nutria and provided justification to remove some of the compounds from the list of acceptable toxicants.

After this test, we did not choose to re-examine bromethalin because of the very poor efficacy (33%) achieved by a 275 ppm bromethalin bait, a concentration 2.75 times higher than that of any other commercial rodenticide, even though bait acceptance appeared sufficient. Although diphacinone tested well with 100% mortality, we were unwillingly to possibly sacrifice efficacy due to nutria consuming

**Table 1.** Active ingredient efficacy trials conducted in December, 2004.

| Nutria Per Test | Challenge Diet (Sweet Potato) | Test Diet (Toxicant-covered Sweet Potato) | | Test Diet Acceptance (%) | Mortality and Duration to Last Death |
|---|---|---|---|---|---|
| | Mean Consumption (g) | Toxicant and Concentration | Mean Consumption (g) | | |
| 9 | 914 | Cholecalciferol (825 ppm) | 475 | 34 | 100% 8 days |
| 9 | 94 | Sodium Monofluoro-acetate (550 ppm) | 173 | 65 | 100% 1 day |
| 9 | 1025 | Bromethalin (275 ppm) | 827 | 45 | 33% 2 days |
| 9 | 1455 | Diphacinone (55 ppm) | 1132 | 44 | 100% 8 days |

vegetation high in vitamin K. Vitamin K is the antidote to diphacinone, and is common in green leafy plants.

Our remaining options were sodium monofluoroacetate and cholecalciferol. Additional active ingredient testing with sodium monofluoroacetate (SM) and cholecalciferol was conducted. Using celery as the carrier, we added 750 ppm cholecalciferol and 500 ppm SM to groups of six animals. We achieved 100% efficacy (i.e., mortality) in both tests. We decided to test how low we could go with the concentration of the SM on celery and still produce mortality. We formulated a 100 ppm SM bait and offered it in a choice test. We achieved mortality in 3.5 hours (184 grams bait consumed). Again, we dropped the concentration and formulated a 10 ppm SM bait and offered it in a choice test. Mortality was achieved in 6 hours (208 grams bait consumed). As a final step, we offered 50 grams of the 10 ppm bait to the last nutria in a no-choice test. Mortality was achieved between 6 (first symptoms) and 23 hours (found dead). Testing with 750 ppm cholecalciferol baits are consistently effective at controlling nutria.

Although the testing with SM appeared to be effective at very low levels, it is highly scrutinized as a vertebrate pest management tool. It was removed from registration as a food bait toxicant in the US in the 1970s. Therefore, it was not considered for further testing, even though our research has shown it to be very effective as a nutria toxicant.

Cholecalciferol was our final choice for a toxicant for nutria control.

**Secondary Toxicity**

Two species were evaluated for secondary toxicity of cholecalciferol. The red-eared slider (*Pseudemys scripta*) was used as a surrogate scavenger to the many species of reptiles that are found in the marshes, and the domestic ferret (*Mustela putorius furo*) was used as a surrogate mammalian predator for the mustelid species found in the marshes.

To prepare the dose for each respective species, five males and five female Norway rats (*Rattus norvegicus*, Wistar strain) were offered 750 ppm bait in a no-choice feeding regime until dead (Table 2). This type of feeding regime would simulate a "worst case scenario" as if an animal solely chose to consume the bait. After the rats had died, they were placed into individually labeled plastic bags and stored in a freezer to minimize any further degradation of the carcass and active ingredient in the carcass. After the rat skin and feet were removed, the carcass was ground in a meat grinder to a medium consistency and then re-ground with the same setting to help mix the carcass. The carcass was then re-ground again with a fine setting and further mixed by hand post-grinding to maintain a homogeneous mix.

*Red-eared Slider.* The turtles were acquired from a local pet store and were transported to Genesis Labs in a cardboard box. Sex and age were unknown, but carapace diameter suggests that they were of adult age.

**Table 2.** No-choice consumption of the cholecalciferol bait by Norway rats.

| Rat ID | Sex | Total Bait Consumption (g) | Total mg/kg Body Weight Consumed | Days until Death |
|---|---|---|---|---|
| 1 | | 66.7 | 97.0 | 3 |
| 2 | | 48.5 | 81.6 | 4 |
| 3 | Male | 8.1 | 18.7 | 9 |
| 4 | | 45.0 | 71.5 | 4 |
| 5 | | 47.3 | 79.2 | 4 |
| 26 | | 60.9 | 187.0 | 3 |
| 27 | | 79.7 | 230.3 | 5 |
| 28 | Female | 70.5 | 196.9 | 5 |
| 29 | | 71.5 | 180.4 | 4 |
| 30 | | 35.6 | 109.4 | 3 |

To deliver the test substance to the turtles, the exit hole of 10 ml syringe was drilled out to allow larger particles to travel through the syringe. A Soverign Feeding Tube and Urethral Catheter (size: 16" 14 Fr.) was cut to 6" with a bevel and attached to the exterior diameter of the syringe neck. The syringe and tube was preloaded with the meat and tared on a balance. Then, an additional amount of meat was added to the syringe and weighed as a pre-dose weight. The turtle was held by a technician and the turtle mouth was kept open with a flat screwdriver, held vertically in the turtle's mouth. The mouth was not pried open, but kept open after the turtle opened its mouth to bite the screwdriver. The tube was slowly placed into the turtle's mouth by rotating the tube and syringe to minimize application resistance of the tube. The syringe plunger was depressed until the test material was observed in the throat or mouth. After dosing the turtle, the syringe was weighed to determine dose. The turtles were held in a plastic tub for 3 hours to assure consumption of the meat and lack of regurgitation.

Upon consultation with Gary Witmer of the USDA National Wildlife Research Center, alligator (*Alligator mississippiensis*) secondary toxicity studies with zinc phosphide suggested that intoxication in reptiles may be postponed instead of the expected quick onset of symptoms. Due to this possibility, our post-treatment observation period was extended.

Table 3 identifies the dosage given to each turtle during the two doses. Female rat #28 (Table 2) was used to dose all four turtles (70.5 g bait consumption, 196.5 mg active ingredient per kg body weight). To produce a worst case scenario, we dosed the turtles on days 0 and 6 with the assumption that a turtle might feed every 6 days within a 12-day maximum exposure period during a field baiting.

During the 20-day post-test observation, no signs were observed that indicated any ill-health of the turtles. This suggests that this species of turtle, and possibly other reptiles, may not be affected by secondary exposures to cholecalciferol from consuming poisoned nutria.

*Domestic Ferret.* The ferrets were acquired from a reputable supplier and were transported to Genesis Labs in wood framed/hardware cloth cages. They were held in plastic-coated wire cages with a floor area of 576 $in^2$ (24 x 24 x 16 in). Ferrets were offered Purina ferret chow *ad libitum*. Bedding pans with wood shavings were placed below each cage and changed at least once each week.

Table 4 identifies the total dosage consumed by each of the two ferrets during the five-day exposure period. The male ferret consumed portions of rat 1, 2, 4, 26, and 29. The female ferret consumed portions of rat 3, 5, 27, 28, and 30 (Table 2).

As an additional measure of intoxication or sub-lethal toxicity, a blood draw was conducted 5 days after the initial exposure for analysis of albumin and calcium levels. Both of the two treatment ferrets and an extra control animal were tested. The control animal was tested to measure normal blood levels. The blood draw results were normal for the control and the treated female, but were elevated with the treated male ferret (Table 5).

Albumin is also measured to identify a concurrent effect of intoxication because albumin is a carrier for calcium. For ferrets, the normal range of calcium

**Table 3.** Total dose of rodent-poisoned Norway rat carcass gavaged to the turtles.

| Turtle ID | Turtle Body Weight (g) | Theoretical Maximum mg a.i./kg body weight (Dec 15, 2005) | Theoretical Maximum mg a.i./kg body weight (Dec 21, 2005) | Total Theoretical Maximum mg a.i./kg body weight |
|---|---|---|---|---|
| 1 | 290 | 2.31 | 3.94 | 6.25 |
| 2 | 268 | 3.23 | 2.50 | 5.73 |
| 3 | 331 | 2.74 | 3.51 | 6.25 |
| 4 | 329 | 1.92 | 2.15 | 4.07 |

**Table 4.** Ferret consumption of the poisoned Norway rat carcass.

| Ferret ID (sex) | Ferret Body Weight (g) | Total Carcass Consumption (g) | Total A.I. Consumed (mg a.i. / kg body weight)[a] |
|---|---|---|---|
| 1 (m) | 1,420 | 905.5 | 322.2 |
| 6 (f) | 814 | 567.4 | 258.3 |

[a] The theoretical consumption is based upon equal distribution of the toxicant in the body and no degradation.

**Table 5.** Ferret blood sample analysis results.

| Ferret ID | Albumin Level (g/dL) | Calcium Level (mg/dL) |
|---|---|---|
| Control | 2.7 | 10.4 |
| Treated Male #1 | 4.1▲ | 18.9▲ |
| Treated Female #6 | 2.8 | 11.5 |

▲ = The hematology instrument indicated elevated levels compared to normal levels.

levels is 8.0 to 11.8 mg/dL. A typical level of calcium that may initiate clinical physical symptoms is ~14 mg/dL. The effects are reversible. Although the male levels indicate an effect, long-term observation (30 days) of the ferrets did not indicate any visible physical effects at any time. During the 25-day post-test observation, no physical signs were observed that indicated any ill-health of the ferrets. This suggests that ferrets may not be affected by secondary exposures to cholecalciferol in nutria carcasses.

## DISCUSSION

Based on this series of research studies, we conclude that 750 ppm cholecalciferol (vitamin $D_3$) bait is an ideal formulation to test further for nutria control.

Although vitamins are essential to survival, large amounts of vitamin $D_3$ can be toxic. Its mode of action involves the mobilization of calcium from the bones into the circulatory system causing hypercalcemia, a highly toxic condition that is not easily reversed. Marshall (1984) reports field studies in which 90 to 98% efficacy was achieved with Norway and roof rats (*R. rattus*) and house mice (*Mus musculus*) in 31 different locations of the US, low toxicological risk to birds (chronic oral $LC_{50}$ quail [*Coturnix coturnix*] 2,000 ppm; mallard [*Anas platyrhyncos*] 4,000 ppm; $LD_{50}$ mallard >2,000 mg/kg), virtually insoluble in water, and low toxicity to fish. A secondary toxicity study conducted with beagles (*Canis familiaris*) resulted in no symptoms and/or death after consuming poisoned rats for 14 consecutive days (Marshall 1984).

The next step in testing is to field test the baits. At this time, pilot field testing will be conducted to test the vegetable-based formula and the gum-based formula. Each bait offers certain positive characteristics that need to be evaluated in a field test. After the pilot field test is conducted on each bait formula, a final formula will be chosen and used in a larger, definitive field test. Other ecotoxicology tests are expected to be conducted as well.

## CONCLUSION

Genesis Laboratories, Inc. has initiated research necessary to develop a floating, efficacious nutria toxicant bait. To date, cholecalciferol is the preferred toxicant, and two candidate formulations are to be tested in a pilot field study. The most successful formulation will be further developed. If another nutria toxicant can be registered, resource managers will have another tool with which to manage the invasive nutria.


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## LITERATURE CITED


CHRISTEN, M. F. 1978. Evalucion nutritive de cuatro dietas monospecificas in la alimentacion del coipo (*Myocaster coypus*). Tesis Faculitad de Medicina Veterinaria, Universidad Austral de Chile, Valdivia.

CONNER, W. H. AND J. R. TOLIVER. 1987. The problem of planting Louisiana swamplands when nutria (*Myocaster coypus*) are present. Proceedings Eastern Wildlife Damage and Control Conference 3:42-49.

EVANS, J. 1970. About nutria and their control. U.S. Fish and Wildlife Service, Resource Publication No. 86.

EVERS, D. E., C. E. SASSER, J. G. GOSSELINK, D. A. FULLER, AND J. M. VISSER. 1998. The impact of vertebrate herbivores on wetland vegetation in Atchafalaya Bay, Louisiana. Estuaries 21:1-13.

FOOTE, A. L. AND L. A. JOHNSON. 1992. Plant stand development in Louisiana coastal wetlands: nutria grazing effects on plant biomass. Wetlands: Proceedings of the Annual Conference 13:265-271.

FORD, M. A. AND J. B. GRACE. 1998. The interactive effects of fire and herbivory on a coastal marsh in Louisiana. Wetlands 18:1-8.

FULLER, D. A., C. E. SASSER, W. B. JOHNSON, AND J. G. GOSSELINK. 1985. The effects of herbivory on vegetation on island in Atchafalaya Bay, Louisiana. Wetlands 4:105-114.

GRACE, J. B. AND M. A. FORD. 1996. The potential impact of herbivory on the susceptibility of the marsh plant *Sagittaria lancifolia* to saltwater intrusion in coastal wetlands. Estuaries 19:13-20.

MACH, J. J. 2004. Rodent bait to control nutria (*Myocaster coypus*). Genesis proposal to USDA Small Business Innovation Research Solicitation # USDA/03-1 Phase I. Award # 2004-00567.

MACH, J. J. 2006. Rodent bait to control nutria (*Myocaster coypus*). Genesis proposal to USDA Small Business Innovation Research Solicitation # USDA/03-1 Phase II. Award # 2006-00310.

TAYLOR, K. L., J. B. GRACE, AND B. D. MARX. 1997. The effects of herbivory on neighbor interactions along a coastal marsh gradient. American Journal of Botany 84:709-715.

TAYLOR, K. L. AND J. B. GRACE. 1995. The effects of vertebrate herbivory on plant community structure in the coastal marshes of the Pearl River, Louisiana, USA. Wetlands 15:68-73.

# GENETIC OPTIONS FOR THE CONTROL OF INVASIVE VERTEBRATE PESTS: PROSPECTS AND CONSTRAINTS

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***Abstract:*** Conventional methods for the control of invasive pests are generally effective only on small-space scales or short-time frames. For most well established pest populations, longer-term efforts to manage the problem have been largely abandoned. I examine the potential of using "autocidal" genetic techniques to control terrestrial vertebrate pests, based on the inheritance through males of transgenes that either sterilise females or convert them into functional males ("daughterless"). Simulation analysis of two high profile pest species, the cane toad (*Bufo marinus*) in Australia and brown rats (*Rattus norvegicus*) in an urban environment, using realistic parameters, suggests that virtual eradication could be achieved at apparently realistic stocking rates within 100 years for toads, and in less than 20 years for rats. The essential genetic requirements for autocidal technology (the ability to genetically transform the pest, genes that when blocked cause sex-specific infertility or sex change, and a means of shutting off the construct for breeding purposes) have already been demonstrated in rodents and are likely to be available in other pests, based on broad conservatism of genetic mechanisms of sex differentiation in vertebrates. Hence, there appear to be no major logistical or technical impediments to developing a genetic control program against many pest species. However, the models also indicate that a recombinant pest control program would be difficult against species whose populations are under strong density dependent regulation or are so large that absolutely high numbers of carriers need to be stocked to achieve control. More potent genetic options than those modelled could be feasible, but their use needs to be tested against public acceptability, due to the apparently higher risk they pose for non-target populations and species.

***Key Words:*** amphibian, cane toad, genetically modified, invasive species, model, Norway rat, policy, public acceptability, recombinant genetics, risk, rodent.

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## INTRODUCTION

Methods to manage invasive species range from ignoring them and hoping they will go away through to a variety of options for physical removal, biocides and biological control (Thresher and Kuris 2004). For well established and widely distributed pests, the only realistic options currently available are augmentative and classical biological control, and sterile male release programs, both of which have significant constraints on their application (Whitten and Foster 1975). As a result, most invasive pests remain uncontrolled except at small scales and for short periods.

In the 1960s, entomologists speculated that genetic techniques could be a powerful means of controlling pest populations (Hamilton 1967), based on the observation that meiotic drive (a genetic sex ratio distorter) had apparently driven some insect populations to extinction. Practical development of such techniques lied fallow, however, until recent developments in recombinant genetics stimulated renewed interest in the field (e.g., Thomas et al. 2000). Currently, at least three recombinant methods for pest control (repressible male sterility, virally vectored immuno-contraception, and female-biased sex ratio distortion) are being tested in the laboratory. Another widely publicised study has speculated that the escape of even one carrier of a "Trojan gene" (a construct that pleiotropically enhances mating advantage while otherwise reducing fitness) could cause species extinction (Muir and Howard 2002), an issue of considerable concern with regards to accidental escapement of genetically-modified (GM) organisms, but also one that offers options for pest control if properly managed. A number of recent studies have modelled the potential for pest control of methods that have been proposed, incorporating varying degrees of ecological reality (Davis et al. 2000, Schlickelman and Gould 2000a,

b, Gould and Schliekelman 2004, Schliekelman et al. 2005, N. Bax and R. Thresher, unpublished data). All conclude that pest control using genetic methods is feasible, at least under the conditions specified in the models.

In this paper, I briefly review the genetic options that have been proposed thus far, discuss their potential when applied to two very different invasive vertebrates (the cane toad [*Bufo marinus*] in Australia and urban populations of the brown rat [*Rattus norvegicus*]), and then consider possible ecological and logistical constraints on the application of the technology for the control of terrestrial vertebrate pests.

## GENETIC OPTIONS FOR PEST CONTROL

Three broadly different approaches for using recombinant genetics to control pests have been investigated: genetically engineered viruses that, when incorporated into a bait, act as a species-specific toxin or sterilizer; self-disseminating engineered viral diseases; and non-disseminating (i.e., sexually transmitted) "autocidal" genes. In the last, I include as a special case chromosomal modifications designed to achieve the same outcomes as the recombinant approaches (Gutierrez and Teem 2006, Cotton and Wedekind 2007). Viruses as toxins will not be considered further here, as their dynamics and efficacy largely parallel those of a conventional baiting program (see Torres et al. 2001 for an example of using a virus to disseminate a recombinant vaccine in rabbits [*Oryctolagus cuniculus*]). Genetically engineered viral diseases are being examined principally for mammals, in which the target is a gene crucial for reproduction that can be disrupted by a suitably modified, otherwise low impact vector (usually a virus). "Immuno-contraception" has been investigated for rabbits, mice (*Mus musculus*), foxes (*Vulpes vulpes*), and cane toads in Australia (Hardy et al. 2006, Robinson et al. 2006) and brushtail possums (*Trichosurus vulpecula*) in New Zealand (Cowan 1996). Despite promising results in the laboratory, a decade long program to develop immuno-contraception against introduced mice in Australia has recently been terminated, in part because of perceived difficulties in obtaining public approval to release the virus and in part because of problems finding a suitable virus.

Gould and Scheikelman (2004) coined the term "autocidal" to refer to genetic modifications that essentially resulted in a species breeding itself to extinction. Five autocidal approaches have thus far been suggested as possible control options (Table 1). All appear to be genetically feasible. Projects to develop or assess the population impacts of four are currently in progress, principally on insects or fish: female-specific lethality (Thomas et al. 2000), "daughterless" and female-specific sterility (Thresher et al. 2005), and a pleiotropic "Trojan" gene (A. Kapuscinski, personal communication). Autocidal techniques are of interest in part because they offer possibilities for control where none now exist (e.g., most invasive pests), but also because they have several intrinsic advantages over conventional biological control: theoretically, at least, and depending on the specific method used, the genes can be constructed to be species-specific; they can target particular life-history stages or sexes, so as to maximise efficacy or minimise damage to non-target species; their effects can potentially be reversed if something goes wrong (using a genetic "off switch" built into the construct); and some approaches lend themselves to relatively quick and inexpensive modification to target other species, while retaining species-specificity for each. The relatively low cost of changing targets spreads the benefits of the high-cost genetic program required to develop an autocidal approach, and contrasts with the virtually "start-all-over-again" efforts needed to find a separate pathogen for each pest species targeted by conventional biological control.

## POTENTIAL APPLICATION TO TERRESTRIAL VERTEBRATE PESTS: EXAMPLE 1 – CANE TOADS

Cane toads were introduced from Hawaii to tropical Australia in 1935, as a biological control agent for cane beetles (Lever 2001). The toad proved singularly ineffective as a biocontrol agent, but rapidly increased its range, population densities and conspicuousness in Australia, to the point where it is considered one of the countries most damaging invasive terrestrial vertebrates, and one of the worlds' 100 worst invasive alien species (Lowe et al. 2001). Currently, the toad occupies about half a million $km^2$ of northern Australia (primarily Queensland and the Northern Territory), but its range is expanding rapidly (Phillips et al. 2006) and it appears that most of tropical and sub-tropical Australia is vulnerable to the species. The toad has been implicated in the decline of native predatory mammals and reptiles, as well as native frogs (Lever 2001), but compelling data of long-

**Table 1.** Autocidal approaches suggested as possible options for controlling invasive pests.

| Approach | Description | References |
|---|---|---|
| Sex or stage-specific lethality/sterility | Construct induces death of offspring at specified stage, or kills or sterilises offspring of one sex, in which case the gene is transmitted through the other sex. | Thomas et al. 2000 |
| Gender distortion ("daughterless" or "sonless") | Construct causes offspring to develop as specified sex irrespective of sexual genotype. | Hamilton 1967, Thresher et al. 2005 |
| Inducible mortality | Construct causes death when externally triggered by, e.g., extreme environmental variability or artificial trigger; construct maintained in population by further stocking. | Grewe 1997, Schliekelman and Gould 2000 |
| Pleiotropy "Trojan gene" | Construct pleiotropically has positive effect on one or more fitness components, and negative effects on others, e.g., increases mating advantage while decreasing viability of genetically modified offspring. | Muir and Howard 1999, 2000 |
| Selfish genes | Operational construct, e.g., one that causes gender distortion or sex-specific lethality, is packaged into a genetic element that has a high probability of reproducing itself within a genome, increasing both its spread and that of the construct. | Burt 2003, Burt and Trivers 2006 |

term impacts on Australian ecosystems are still sparse (see papers in Molloy and Henderson 2006). The history of efforts to control the toad in Australia is reviewed by Tyler (1998) and Lever (2001). A variety of methods have been suggested to control it, ranging from topical biocides to traps, sterile male release programs and classical biological control (see papers in Molloy and Henderson 2006). To date, no safe and effective biological control agent against the species has been identified, and other approaches have proven successful only at small scales.

The possible effects of a recombinant program of pest control on cane toads have been examined by Thresher and Bax (2006), the results of which are summarised here. The assessment was made using a deterministic, age-structured population genetics model that incorporated the main features of cane toad demography, based on information from Lampo and De Leo (1998) and R. Shine (persoanl communication). The population genetics model simulates recruitment, age-specific mortality, sex ratios and gene frequencies in a freely interbreeding population; i.e., it assumes no emigration and immigration. Initial conditions for the cane toad specify an annual mean recruitment of 1,000 individuals at carrying capacity, a sex ratio at birth of 1:1, a constant post-metamorphic mortality rate, an age at maturity (95% mature) of 2 years, and a maximum age (95% mortality) of 5

years. Recruitment (number of individuals surviving metamorphosis) was related to adult population size by means of a discrete logistic (Ricker) model. Lampo and De Leo (1998) suggest strong density dependence in the egg and larval stage, but only slight density dependence among juveniles and adults. This was captured in the model by using as a baseline a Ricker parameter of 1.25, which specifies low recruitment to the juvenile stage when populations are at less than half carrying capacity (due to the small number of breeding females) but which stabilises asymptotically at larger population sizes due to competition among pre-metamorphs. The model also assumes recruitment is strongly affected by rainfall (proxied as the Southern Oscillation Index), on the basis that high water levels increases the number of available breeding ponds and resources for developing tadpoles. Extinction is arbitrarily defined as when the number of viable females falls to 1% of initial population size, and the model runs are limited to a maximum of 100 years, as any method that took longer than this to control cane toads would probably not be attractive to managers or the public.

The effects of physical removal on the modelled toad population are shown in Figure 1, for removal levels of up to 20% of the breeding population each year. Physically removing toads reduces immediate population density, obviously, but at low removal rates this is largely over-ridden by large inter-annual differences in breeding success and by density dependence. To achieve a significant long-term effect on population size, about 20% of the adults need to be removed each year; long-term extinction requires annual removal in excess of 40% – a huge task.

Figure 2 compares the effect of four recombinant options on cane toad numbers: a gene construct that biases offspring sex ratios towards males ("daughterless"), one that sterilises females, but not males ("female sterile"), one that causes pre-maturational female-specific mortality ("female lethal"), and a Trojan gene. For the last, the mating success of male carriers was set 4 times higher than that of wild type males, but they also had 0.5% lower first-year survival of their offspring, a combination suggested by Muir and Howard (2002) as most likely to lead to population extinction. For the comparison, stocking rate was set at 5% of mean annual pre-stocking recruitment per year, and carriers were assumed to have eight

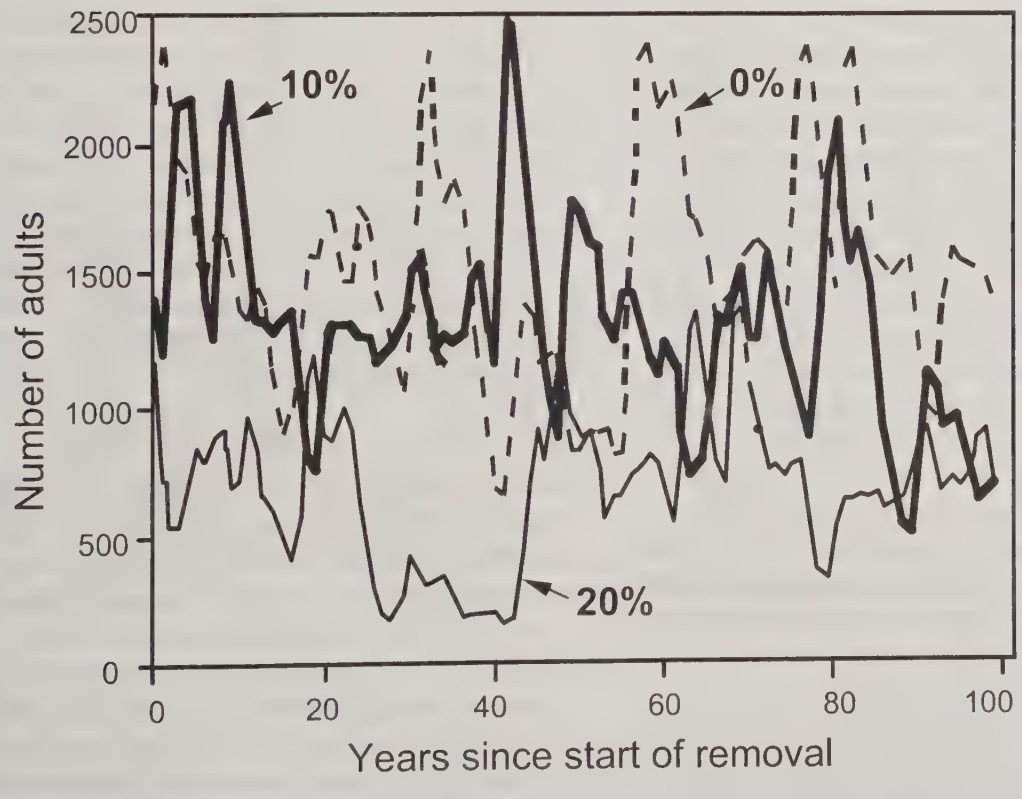



**Figure 1.** Effects of physical removal on modelled number of cane toads, at annual removal rates of 0, 10 and 20% of the post-metamorph population.

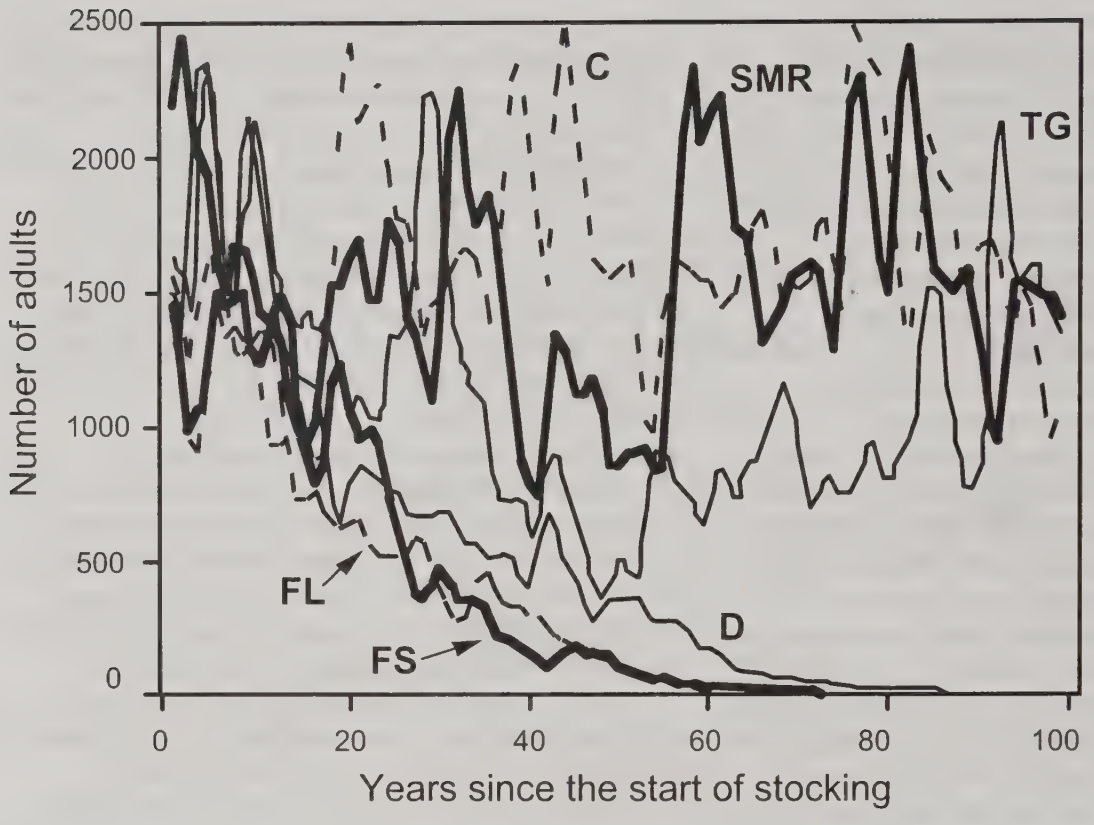



**Figure 2.** The effect of six genetic control methods on numbers of adult cane toads. C = control (stocking of neutral gene); SMR = Sterile male release; TG = Trojan gene; FL = Female-specific lethality; FS = female-specific sterility; D = daughterless.

independently segregating copies of the relevant construct. The model also assumes that one copy of the construct was sufficient to cause the desired phenotype (sex change, sterility or death), that there was no leakage due to gene silencing, and that the stocked animals had a fitness equal to that of the wild-types. The effects of varying these parameters on the efficacy of a recombinant program are explored by N. Bax and R. Thresher (unpublished data). For completeness, the model also assesses the effects on the cane toad of stocking a neutral gene (a stocking control) and the release of sterile males (also at 5%) as a possible alternative management option.

At the 5% per annum stocking rate, neither releasing sterile males nor carriers of a Trojan gene had a significant long-term effect on toad numbers. The model suggests that achieving extinction using a sterile male release program requires annual releases in excess of 50% of the number of wild type males and even then, takes about 90 years of annual stocking. The Trojan gene method results in a rapid replacement of the wild genotype (N. Bax and R. Thresher, unpublished data), but does not lead to population extinction even at stocking rates as high as 70% of mean pre-stocking recruitment. In contrast, 5% annual stockings of carriers of female-lethal, female-sterile or daughterless constructs cause the toad population to go extinct in less than 80 years. The minimum annual mean stocking rates required to achieve extinction within 100 years are similar for the female-lethal and female-sterile constructs (at about 2.5% of natural recruitment levels) and is slightly higher for a daughterless construct (3%).

## POTENTIAL APPLICATION TO TERRESTRIAL VERTEBRATE PESTS: EXAMPLE 2 – THE NORWAY RAT

Urban rodents are among the world's most damaging invasive species as disease vectors and threats to public health, contaminators and consumers of food products, and destroyers of infrastructure (Pimental et al. 2000, Sullivan 2004). Despite this, and despite an almost universal dislike of urban rats, in particular, most urban governments have effectively abandoned efforts to control mice and rats at anything other than local scales and over short periods, using poisons, traps, barriers and

acoustic devices to lure, repel or, most frequently, kill rats and mice. The long-term effects of these control efforts are negligible (Davis 1987, Easterbrook et al. 2005), which has led to calls for better means of managing the pests (Colvin and Jackson 1999).

To assess the potential of a recombinant approach to control invasive rodents, I use as a test case an urban population of the Norway or brown rat. The model structure and key assumptions are similar to that described above for the cane toad. Basic parameters for the Norway rat model are based on Davis (1953): sex ratios are equal at birth, average age to maturity is 3.5 months, 99.9% of the newborn die before or at weaning (annualised mortality rate), the subsequent annual mortality rate is 91%, and recruitment is density dependent and strongly affected by environmental variability. As for cane toads, I assume as a first approximation that the rat population is closed; this assumption seems reasonable based on field data showing that Norway rats are relatively non-migratory in urban environments (Davis 1953).

The effect of introducing multiple copies of a female-sterilising construct, inherited through males, on a Norway rat population the size of Baltimore's (approximately 50,000 rats) (Easterbrook et al. 2005) is shown in Figure 3, with and without complementary rat control. Alone, releasing adult carriers equivalent to 5% of wild-type (WT) recruitment at carrying capacity (approximately 1,750 rats per month, of which half are male carriers and the rest sterile females) reduces the rat population to virtual extinction in less than 20 years. Combining the 5% monthly stocking with a control program that removes genetically modification and WT rats indiscriminately at a rate that doubles the natural mortality of the adults reduces the modelled rat population by more than 99.9% in about 10 years. In contrast, just increasing the rat mortality without the genetic modification technology only depresses rat numbers without eradicating them (Figure 3), and the population quickly rebounds if the killing stops (Davis 1987). Producing about 2,000 rats per month to release requires a significant effort, but

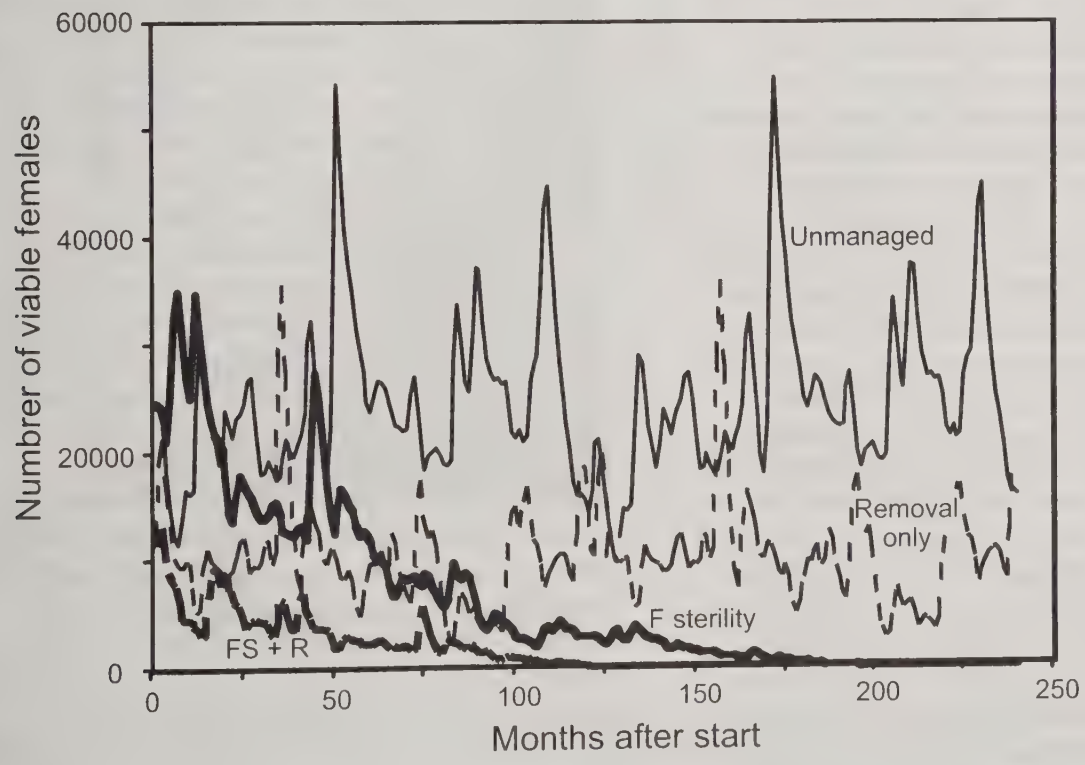



**Figure 3.** Modelled trajectories for an urban rat population under four management conditions: unmanaged, sustained annual removals/killings that double the rate of adult natural mortality (removal only), a female-sterility construct stocked at 5% of natural pre-treatment recruitment (F sterility), and a combination of both the recombinant method and non-discriminate removal of rats (FS + R), as above.

appears to be feasible given the scale of the commercial rat breeding industry. The ease with which rats can be bred suggests options for community involvement in a "breed-and-release" program, which could also build local interest in and support for the program.

## GENETIC FEASIBILITY OF RECOMBINANT CONTROL METHODS

The model analyses indicate that (1) physical removal (killing, trapping, etc.) alone is insufficient to control either well-established invasive vertebrate – cane toads or rats – at anything approaching realistic effort levels, (2) recombinant methods can achieve control, and possibly eradication, within what are likely to be acceptable time frames and at effort levels that are likely to be costly, but achievable, and (3) combining the recombinant method with physical removal reduces the time required to control the pest. The limited impact of physical removal as a pest control option with some species is well established. Nonetheless, the analyses indicate that it can play a significant role in a pest management program based on recombinant methods. In an idealised scenario, recombinant approaches can be considered "silver bullets" for pests, but they work best when combined with other pest control options in an integrated pest management framework. The modelled control programs are based on a construct inherited through one sex (males) that sterilises, kills or changes the sex of the other (females). As such, they require the ability to genetically transform the target species, genes that when knocked-out or mis-expressed cause the phenotype of interest in females, and a method for repressing the effect of the construct(s) in order to maintain breeding lines (conditional gene expression). The extent to which these requirements are satisfied at present varies widely across taxa, depending on the amount of genetic interest in them for other purposes. Hence, existing technologies appear to satisfy all three requirements in rodents: transgenic rats and mice have been routinely produced for medical research since the 1990s (Tesson et al. 2005), numerous genes that when disrupted cause sex-specific infertility in rats and mice have been described (e.g., Al-Shawi et al. 1992, Sharpe et al. 1995), and conditional gene expression, using the Tet-off system (Gossen et al. 1995), has been demonstrated in mice (Kistner et al. 1996). In contrast, considerably less background work has been done on amphibians, in general, and toads in particular. There are no reports in the literature of genetic modification of toads, but there is a large literature on the use of sperm-mediated gene transfer to produce transgenic frogs (Ny et al. 2006). The African clawed frog (*Xenopus laevis*) is a model species for studying the genetics of development in lower vertebrates (e.g., Heasman 2006), and hence techniques for manipulating gene expression in the species have been well worked out. It is highly likely that the methods used to produce transgenic tadpoles and adult *X. laevis* would be broadly applicable to cane toads. With regard to suitable genes, a general program to develop at daughterless construct in cane toads is described by Koopman (2006). More specifically, development of a daughterless construct in fish has focused on disrupting the expression of the enzyme aromatase, which is crucial in female sexual differentiation (Thresher et al. 2005, in preparation). Chemical inhibition of aromatase modifies phenotypic sex in amphibians (Chardard and Dournon 1999, Mackenzie et al. 2003), as it does in fish, suggesting that an approach similar to that being developed for fish would be effective in the toads.

More broadly, patterns of steroid and hormone expression and activity are similar across a range of vertebrate lineages (Devlin and Nagahama 2002, Eggert 2004), as are many elements of the underpinning genetics (Devlin and Nagahama 2002, Koopman 2006, Wilhelm and Koopman 2006). This conservatism should facilitate identification of suitable target genes in novel species. Again, aromatase provides a possibly useful example. The enzyme serves a similar function in most lower vertebrate lineages, has been sequenced from taxa as diverse as fish, reptiles and mammals (including humans), and when blocked chemically, sex reverses genetic females into functional males in fish, amphibians, reptiles and birds (Elbrecht and Smith 1991, Piferrer et al. 1994, Wibbels and Crews 1994, Charnaud and Dournon 1999). The ubiquity of the enzyme and its key role in sex determination across lower vertebrates suggest it to be a good candidate gene on which to base sex-specific pest control options. Similarly, methods for genetic modification appear to have wide applicability, with a range of commonly used techniques, such as micro-injection, sperm- or virally-vectored transfection, demonstrated in fish, a frog, a diverse range of mammals (including rabbits, cows [*Bus taurus*], pigs [*Sus scrofa*], cats [*Felis catus*], sheep [*Ovis aries*], and goats [*Capra hircus*]) (Houdebine 2005) and, recently, a bird

(Koo et al. 2006). The Tet-off/Tet-on system of conditional gene expression also appears to be widely applicable across vertebrate lineages, having been demonstrated in animals ranging from fish to small mammals. Hence, although there will be non-trivial challenges to implementing a recombinant program in each new target species, generic approaches and information suggest the principal constraints are likely to be logistical, rather than conceptual. A possible exception are those species that lay eggs, as conventional applications of methods to transform species are greatly facilitated by access to the developing egg and embryo. Genetic transformation has recently been reported in the chicken (Koo et al. 2006) using a retroviral vector, but whether this approach can be safely applied to other egg-laying vertebrates, such as invasive Burmese pythons (*Python* spp.), is not clear.

## ECOLOGICAL CONSTRAINTS ON RECOMBINANT APPROACHES

The model analyses also indicate several features of autocidal control programs that constrain their utility. The first is that like any control program, they perform poorly when the targeted pest population is under strong density dependent regulation (N. Bax and R. Thresher, unpublished data) (Figure 4A). If population regulation is density independent, then any control strategy that consistently reduces the number of breeding adults will have a cumulative, negative effect on population size. For most species, an assumption of density independent population regulation is not easily justified. Estimation of the degree of density dependence in natural populations is difficult (Sibly et al. 2005), but recent meta-analyses (Brook and Bradshaw 2006) suggest

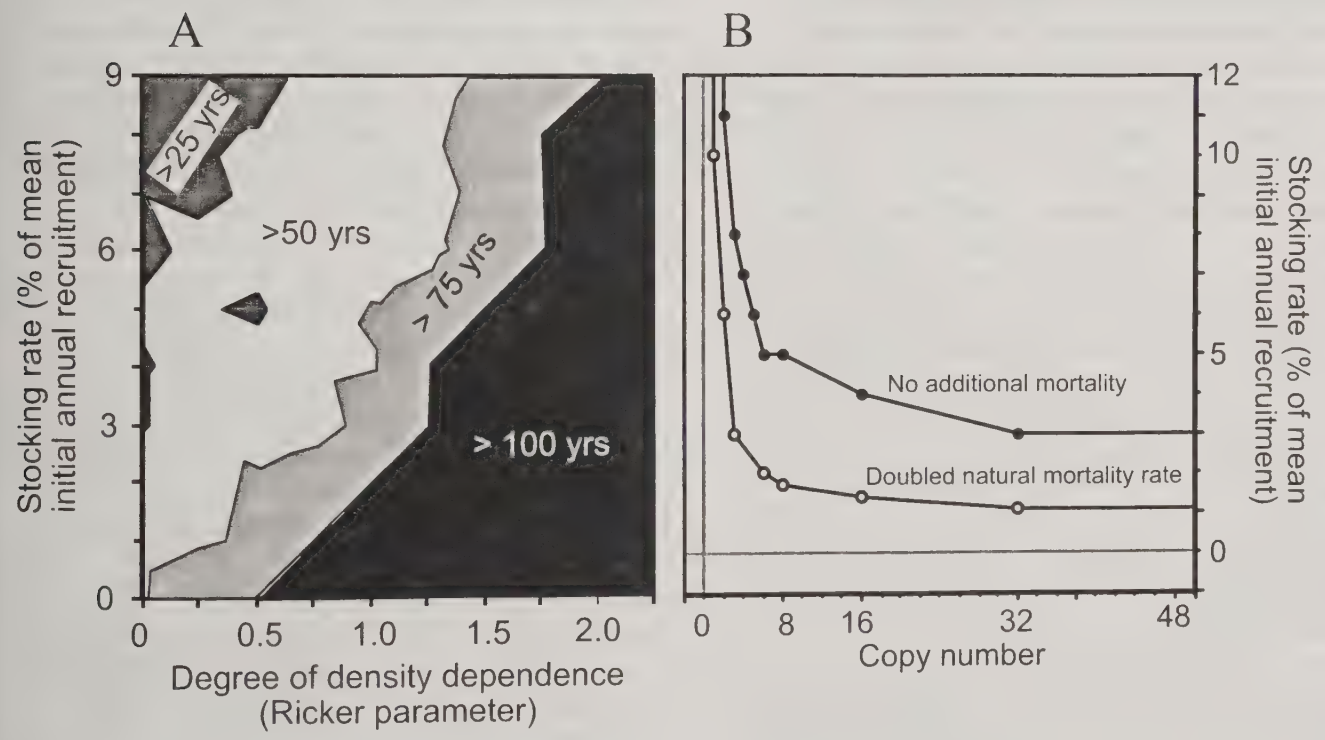



**Figure 4 (A.)** The effect of different degrees of density dependence on the stocking rate required to reduce a cane toad population to <1% of virgin biomass within 100 years, based on the model depicted in Figs. 1 and 2. (**B.**) The relationship between the stocking rate and copy number required to drive a 50,000 rat population to <50 rats within 20 years, based on the model depicted in Figure 3, with and without complementary rat management activities. The slight irregularities in the relationships reflect stochastic effects of environmental variability.

widespread evidence of density dependence across almost all taxonomic groups for which long time-series of population data were available. Density dependence slows and potentially stops the fall in pest numbers as control efforts are maintained, an effect that can be countered only with increasing difficulty by increasing control efforts (Figure 4A).

The second conspicuous feature of autocidal control programs is that they are inherently slow acting. Even under optimal conditions, effective population control (e.g., populations reduced to < 1% of virgin biomass) typically requires more than 10 generations, and can take much longer. On the positive side, this slow impact allows an affected ecosystem time to adjust to the absence of the pest, there are no mass mortalities of pests, and if a problem develops, there is adequate time to launch counter-measures, such as a second gene construct that effectively shuts off the first. On the negative side, managers and the public are likely to be unimpressed with, and may not be prepared to invest in, such a slow response to the problem. For urgent problems, such as when a native species is at risk, a genetic approach may be too slow to be useful. More fundamentally, the slow action of a recombinant pest program could allow time for the evolution of strongly counter-selected mechanisms to, for example, re-balance operational sex ratios. Logically, as sex ratios in a daughterless program become increasingly skewed towards males, those animals able to continue to produce female progeny would have a very high fitness and be selected for. Such rapid genetic responses to distorted sex ratios have been reported in wild populations (Charlat et al. 2007). Whether or not it can occur, or would occur frequently, in vertebrate pest populations can only be determined empirically. If it is a problem, countering it is likely to require simultaneous application of multiple approaches to pest control (integrated pest management) and possibly a suite of autocidal constructs, that could be sequentially released into a target population to over-come counter-selective pressures.

The amount of time required to control the pest problem depends in part on management objectives (complete eradication or a sustained reduction in pest numbers/impacts) and effort expended. For the modelled approaches to work effectively, minimum stocking rates equivalent to 3-5% of annual mean natural recruitment are required, sustained for at least 10-20 generations, to achieve virtual pest extinction; lower stocking rates can depress pest numbers under most conditions, but do not result in extinction. Achieving a stocking rate equivalent to 5% of natural recruitment is not a trivial task for any highly successful invasive species. The scale of the effort depends in part on the biology of the pest, and particularly the ease with which it can be bred, in part on the size of the target population (and hence the absolute number of carriers required), and in part on the extent to which other management factors can be manipulated to optimise effects of stocking (N. Bax and R. Thresher, unpublished data). The efficacy of a stocking program is determined by three factors amenable to management action: stocking rate itself, copy number, and the amount and specificity of complementary efforts to remove the pests.

The interaction of these factors is illustrated for urban rats in Figure 4B. Copy number is the number of independently segregating copies of the construct carried by each stocked animal. The higher the copy number, the larger the number of generations over which offspring of the carriers will express the desired phenotype (Gould and Schliekelman 2004). In principal, at an infinite copy number the release of a single carrier will drive a pest species to extinction, as the proportion of carriers in the population increases geometrically with each subsequent generation. In practice, (1) genetic mechanisms silence redundant copies of genes, in plants at least (Schubert et al. 2004), and (2) the interaction between copy number and stocking rate is non-linear (N. Bax and R. Thresher, unpublished data) (Figure 4B). Hence, with no complementary management activity, a rat population the size of Baltimore's could be virtually eradicated in less than 20 years by releasing 42 rats (21 carrier males and 21 sterile females) per month, 1.2% of natural recruitment, if each had an infinite copy number, 4% per month for a copy number of 16, and 5% at a copy number of 8. If physiological and genetic silencing mechanisms constrain the carriers to be homozygous at a single locus for the construct (copy number = 2), eradication within 20 years is still possible, but requires a stocking rate of 11% per month. Adding non-selective removal (killing or trapping) of rats as part of an IPM program, however, substantially reduces the stocking rate required to achieve eradication. Doubling the rate of natural mortality of post-weaning juvenile and adult rats reduces the required monthly stocking rate of copy number 2 carriers to only 4%, results in virtual extinction within 20 years even with single copy carriers, at a stocking rate of 10%, and

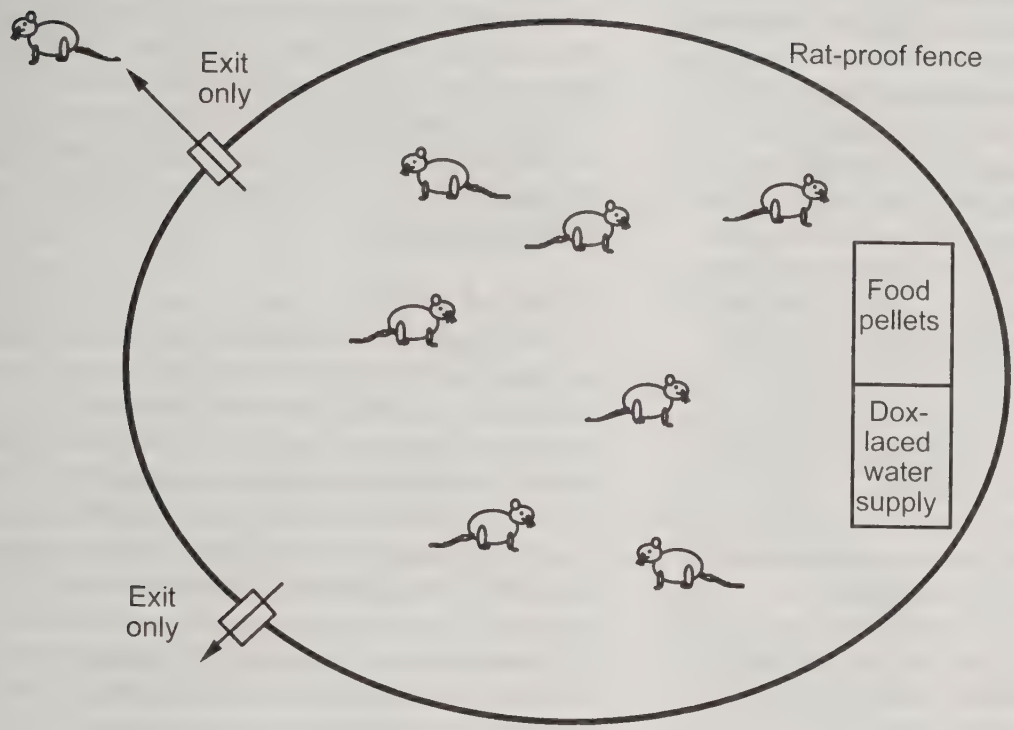



**Figure 5.** Diagram of a self-sustaining seed population of rats, that could be deployed to control island populations. The enclosure is seeded initially with a pair of high copy number rats, with closed exits, a food supply and a water supply dosed with doxyclycline, which represses the female sterility construct and hence allows the pair to breed and numbers in the enclosure to build. At intervals, the exits are opened, allowing some juvenile carriers to escape and spread the autocidal gene to the target population. The system, as diagrammed, would be largely self-contained, requiring only occasional visits to add food and doxycycline supplies.

makes possible eradication at stocking rates of <2% for copy numbers ≥ 6.

The interactions between stocking rate, copy number and complementary management actions on the efficacy of a control program have three implications.

First, complementary management activity enhances a pest control program based on recombinant methods. Selective removal of wild type animals, while leaving the genetically modified carriers to breed, is the most effective approach (N. Bax and R. Thresher, unpublished data), but even non-selective removal can significantly reduce the time to extinction or the stocking rate required. Recombinant methods add value to physical removal programs that otherwise would have little long-term impact.

Second, the interactions imply that recombinant approaches, as they are outlined above, are probably applicable only to pest animals that are present in relatively small numbers (10s to 100s of thousands), that inhabit discrete habitat patches each of which is treated independently, or that can be efficiently and inexpensively bred and stocked out in large numbers. Hence, urban rats rather than free-ranging nutria (*Myocastor coypus*). As many pest species are highly fecund, large-scale breeding programs could often be feasible. Combining the autocidal construct with a conditional regulatory element; i.e., a sequence that turns a gene on or off in response to an environmental stimulus, offers potential for the development of breeding colonies of high copy number individuals, that can seed surrounding areas. Again using rodents as an example, a Tet-off regulatory sequence shuts down gene expression when carriers are provided drinking water dosed with tetracycline or one of its derivatives (Kistner et al. 1996). Hence, a seed population of high copy number rats could consist of a semi- enclosed colony that is provided food and a tetracycline-dosed water supply, from which most, but not all juveniles are allowed to escape (Figure 5). When the escaped carriers breed with wild type rats, the autocidal gene expresses in their

offspring, leading to its spread and a long-term population decline. On small scales, self-sustaining breeding enclosures like this could control rats on islands where biocidal approaches are not feasible or risk damage to high value non-target species. On larger scales, aggressive stocking so as to genetically fix the construct in source regions of a meta-population, which are then sustained by conditional repression as long as is required, could constitute an inexpensive, large-scale source of carriers that either disperse naturally or can be stocked into other areas. In such cases, continuous conditional repression of the construct may not be required for species in which the original breeding adults are long-lived, e.g., pythons.

The third implication is that for pests that are very abundant or in which production of carriers is costly, more aggressive genetic approaches may be required to affect control. Three options have been suggested. First, if silencing prevents production of high copy number individuals, the same effect could be achieved by the simultaneous use of multiple autocidal constructs. In rodents, the diversity of genes known to affect fertility appears to provide considerable scope for multiple constructs, each based on disrupting or mis-expressing a different gene and each subject to its own constraints on copy number. The combination of 4 autocidal constructs, even if each is individually constrained, could result in an effective high copy number, minimising required stocking rates. Second, a single autocidal construct could be packaged in a "selfish" gene element, so that it replicates within the genome and thereby spreads as if present at an infinite copy number (Burt 2003). Options for achieving a self-replicating autocidal gene are discussed by Burt and Trivers (2006), but thus far it remains only a theoretical possibility. Finally, as noted above, it has been suggested that the release of a single carrier of a pleiotropic Trojan gene, i.e., one that increases male reproductive attractiveness while decreasing the viability of its offspring, could lead inevitably to population extinction (Muir and Howard 1999, 2002). More recent analyses suggest extinction is very unlikely under most ecologically realistic scenarios (N. Bax and R. Thresher, unpublished data), but the broader concerns about aggressive genetic approaches raised by Muir & Howard (1999, 2002) remain valid. Whereas a construct that can cause population collapse following the release of a handful, or even one carrier is superficially attractive, it can also constitute a serious threat to non-target species and populations. The use of low copy number carriers necessitates a substantial stocking program to affect the target population, which while logistically costly, makes it unlikely that the accidental release of a few carriers in areas where they are not wanted, or that the spread of a single autocidal construct across a species barrier, would cause long-term damage.

## CONCLUSION

There appear to be no major logistical or technical impediments to use recombinant technology as a core element of a program to control at least some vertebrate pests, such as rats in urban environments, though the required constructs need to be assembled and tested for functionality and fitness effects. Such programs could have major impacts on environmental amenity, biodiversity conservation, health, and even sanitation, as well as significant economic benefits. However, the viability of such a program also depends critically on whether the public accepts the approach (Gaskell et al. 1999) and on a policy and legislative framework that effectively regulates environmental applications of GM technology (Kapuscinski and Patronski 2005). Australian studies suggest that the extent of public acceptability will depend on whether or not the public is consulted during the development of the technology, the transparency of the decision-making process leading to its potential use, and the risk of the construct jumping to non-target species (Thresher and Kuris 2004, Fisher and Crib 2005). Genetic threats to other species need to be assessed on a case-by-case basis, as it is likely to depend on the particulars of the autocidal construct involved and whether or not the target species co-occurs with related species. Currently, in most jurisdictions, autocidal programs will also have to cope with policy and legislative frameworks that provide few clear mechanisms for evaluating and regulating GM technology for purposes of environmental remediation. In the United States, for example, numerous overlapping international, federal and state legislations could affect program viability and it is very ambiguous even as to which regulatory agencies have primary carriage of the issue (Kapuscinski and Patronski 2005). At the other end of the spectrum, Australia has established a national Office of the Gene Technology Regulator, which in principle provides a single point of contact for developing, evaluating and potentially approving environmental release of an autocidal construct. As

the release of the GM carriers constitutes a form of biological control, regulatory frameworks based on those in place for biological control may often be appropriate, suitably modified to encompass expert input into the risk elements specific to recombinant genetics. Unfortunately, regulatory frameworks for biological control in some jurisdictions also appear to be confusing and inconsistently applied (Strong and Pemberton 2000).

These observations suggest that the development of a program to control urban rats, cane toads or any other vertebrate pest using recombinant genetics should involve three initiatives: (1) a program of public consultation to determine the likely acceptability of the approach and the conditions under which it would be acceptable, (2) a review and rationalisation of relevant legislation, possibly in the context of broader issues associated with regulation of biological control in general, and (3) development of the technology itself. The last should include assembly of the relevant constructs and production of integrated lines, evaluation of effects on male fitness in particular, and the development of predictive models that incorporate detailed information on behavior, ecology, demography and inter-specific interactions. If results of these initiatives are positive, it may be practical to field test the genetics and models on invasive populations at physically isolated locations, such as rodents on isolated islands, a major conservation challenge in its own right, at minimum risk and under conditions in which the GM carriers could be eradicated using conventional methods, if need be (Veitch and Clout 2002). Successful tests on small isolated populations could be an important step in developing community and management support for wider application of the technology.

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## LITERATURE CITED

AL-SHAWI, R., J. BURKE, J. J. MULLINS, R. M. SHARPE, R. LATHE, AND L. MULLINS. 1992. Transgenesis and infertility. Pages 195-206 *in* S. G. Hillier, editor. Gonadal development and function. Sereno Symposium. Raven. New York, USA.

BROOK, B. W., AND C. J. A. BRADSHAW. 2006. Strength of evidence for density dependence in abundance time series of 1198 species. Ecology 87:1445-1451.

BURT, A. 2003. Site-specific selfish genes as tools for the control and genetic engineering of natural populations. Proceedings of the Biological Society of London 270:921-928.

BURT, A., AND R. TRIVERS. 2006. Genes in conflict. Harvard University, Cambridge, Massachusetts, USA.

CHARLAT, S., E. A. HORNETT, J. H. FULLARD, N. DAVIES, G. K.RODERICK, N. WEDELL, AND G. D .D. HURST. 2007. Extraordinary flux in sex ratio. Science 317:214.

CHARNARD, D., AND C. DOURNON. 1999. Sex reversal by aromatase inhibitor treatment in the newt *Pleurodeles waltl*. Journal of Experimental Zoology 283:43-50.

COLVIN, B. A., AND W. B. JACKSON. 1999. Urban rodent control programs for the 21ST century. Pages 243-257 *in* G. R. Singleton, L. A. Hinds, H. Leirs and Z. Zhang, editors. ACIAR Monograph No. 59.

COTTON, S., AND C. WEDEKIND. 2007. Control of introduced species using Trojan sex genes. Trends in Ecology and Evolution 22:441-443.

COWAN, P. E. 1996. Possum biocontrol: prospects for fertility regulation. Reproductive Fertility and Development 8:655-660.

DAVIS, D. E. 1953. The characteristics of rat populations. Quarterly Review of Biology 28:373-401.

DAVIS, D. E. 1987. Early behavioural research on populations. American Zoologist 27:825-837.

DAVIS, S. A., E. A. CATCHPOLE, AND G. R. FULFORD. 2000. Periodic triggering of an inducible gene for control of a wild population. Theoretical Population Biology 58:95-106.

DEVLIN, R. H., AND Y. NAGAHAMA. 2002. Sex determination and sex differentiation in fish: an overview of genetic, physiological and environmental influences. Aquaculture 208:191-364.

EASTERBROOK, J. D., T. SHIELDS, S. L. KLEIN, AND G. E. GLASS. 2005. Norway rat population in Baltimore, Maryland, 2004. Vector-borne Zoonotic Diseases 5:296-299.

EGGERT, C. 2004. Sex determination: the amphibian models. Reproduction and Nutritional Development 44:539-549.

ELBRECHT, A., AND R. G. SMITH. 1991. Aromatase enzyme activity and sex determination in chickens. Science **255**:467-470.

FISHER, N., AND J. CRIBB. 2005. Monitoring community attitudes to using gene technology methods (daughterless carp) for managing common carp. Report prepared for the Cooperative Research Centre for Pest Animal Control, Valuemetrics Australia.

GASKELL, G., M. W. BAUER, J. DURRANT, AND N. C. ALLUM. 1999. Worlds apart? The reception of genetically modified foods in Europe and the U.S. Science 285:384-387.

GOSSEN, M., S. FREUNDLIEB, G. BENDER, G. MULLER, W. HILLEN, AND H. BUJARD. 1995. Transcriptional activation by tetracycline in mammalian cells. Science 268:1766–1769.

GOULD, F., AND P. SCHLIEKELMAN. 2004. Population genetics of autocidal control and strain replacement. Annual Review of Entomology 49:193-217.

GREWE, P. 1997. Potential of molecular approaches for the environmentally benign management of carp. Pages 119-127 *in* J. Roberts, and R. Tilzey, editors. Controlling carp: exploring the options for Australia. Murray-Darling Basin Comm., Canberra, Austraila.

GUTIERREZ, J. B., AND J. L. TEEM. 2006. A model describing the effect of sex-reversed YY fish in an established wild population: the use of a Trojan Y chromosome to cause extinction of an introduced exotic species. Journal of Theoretical Biology 241:333-341.

HAMILTON, W. D. 1967. Extraordinary sex ratios. Science 156:477-488.

HARDY, C. M., L. A. HINDS, P. J. KERR, M. L. LLOYD, A. J. REDWOOD, G. R. SHELAM, AND T. STRIVE. 2006. Biological control of vertebrate pests using virally vectored immunocontraception. Journal of Reproductive Immunology 71:102-111.

HEASMAN, J. 2006. Patterning the early *Xenopus* embryo. Development 133:1205-1217.

HOUDEBINE, L. M. 2005. Use of transgenic animals to improve human health and animal production. Reproduction in Domestic Animals 40:269-281.

KAPUSCINSKI, A. R., AND T. J. PATRONSKI. 2005. Genetic methods for biological control of non-native fish in the Gila River system. Contract report to the U.S. Fish and Wildlife Service. Minnesota Sea Grant Publication F20. University of Minnesota, Institute of Social, Economic and Ecological Sustainability, St. Paul, Minnesota, USA.

KISTNER, A., M. GOSSEN, F. ZIMMERMAN, J. JERECIC, C. ULLMER, H. LUBBERT, AND H. BUJARD. 1996. Doxycycline-mediated quantitative and tissue-specific control of gene expression in transgenic mice. Proceedings of the National Academy of Sciences 93:10933-10938.

KOO, B. C., M. S. KWON, B. R. CHOI, J. H. KIM, S. K. CHO, S. H. SOHN, E. J. CHO, H. T. LEE, W. CHANG, I. JEON, J. K. PARK, J. B. PARK, AND T. KIM. 2006. Production of germline transgenic chickens expressing enhanced green fluorescent protein using a MoMLV-based retrovirus vector. FASEB Journal 20:2251-2260.

KOOPMAN, P. 2006. Daughterless cane toads. Pages 111-116 *in* K. Molloy, and W. Henderson, editors. Science of cane toad invasion and control. Proceedings of the Invasive Animals CRC/CSIRO/Qld NRM&W Cane Toad Workshop, June 2006. Brisbane, Australia. Invasive Animals Cooperative Research Centre, Canberra, Australia.

LAMPO, M., AND G. A. DE LEO. 1998. The invasion ecology of the toad *Bufo marinus* from South America to Australia. Ecological Applications 8:388-396.

LEVER, C. 2001. The cane toad: the history of a successful coloniser. Westbury Academic and Scientific, Otley, West Yorkshire, United Kingdom.

LOWE, S., M. BROWNE, S. BOUDJELAS, AND M. DEPOORTER. 2001. 100 of the world's worst invasive alien species, a selection from the Global Invasive Species Database. IUCN-ISSG, Aukland, New Zealand.

MACKENZIE, C. A., M. BERRILL, C. METCALFE, AND B. D. PAULI. 2003. Gonadal differentiation in frogs exposed to estrogenic and antiestrogenic compounds. Environmental Toxicology and Chemistry 22:2466-2475.

MOLLOY, K., AND W. HENDERSON. 2006. Science of cane toad invasion and control. Proceedings of the Invasive Animals CRC/CSIRO/Qld NRM&W Cane Toad Workshop, June 2006. Brisbane. Australia. Invasive Animals Cooperative Research Centre, Canberra, Australia.

MUIR, W. M., AND R. D. HOWARD. 1999. Possible ecological risks of transgenic organism release when transgenes affect mating success: sexual selection and the Trojan gene hypothesis. Proceedings of the National Academy of Sciences 96:13853-13856.

MUIR, W. M., AND R. D. HOWARD. 2002. Assessment of possible ecological risks and hazards of transgenic fish with implications for other sexually reproducing organisms. Transgenic Research 11:101-114.

NY, A., M. AUTIERO, AND P. CARMELIET. 2006. Zebrafish and *Xenopus* tadpoles: small animal models to study angiogenesis and lympangiogenesis. Experimental Cell Research 312:684-693.

PHILLIPS, B. L., G. P. BROWN, J. WEBB, AND R. SHINE. 2006. Runaway toads: an invasive species evolves speed and thus spreads more rapidly through Australia. Nature 439:803.

PIFERRER, F., S. ZANUY, M. CARRILLO, I. SOLAR, R. H. DEVLIN, AND E. DONALDSON. 1994. Brief treatment with an aromatase inhibitor during sex differentiation causes chromasomally female salmon to develop as normal, functional males. Journal of Experimental Zoology 270:255--262.

PIMENTAL, D., L. LACH, R. ZUNIGA, AND D. MORRISON. 2000. Environmental and economic costs of non-indigenous species in the United States. Bioscience 50:53-65.

ROBINSON, T., N. SIDDON, S. TARMO, D. HALLIDAY, T. SHANMUGANATHAN AND D. VENABLES. 2006. CSIRO biocontrol project: concept and progress. Pages 86-88 *in* K. L. Mollay, and W.R. Henderson, editors. Science of cane toad invasion and control. Invasive Animals CRC, Canberra, Australia.

SCHLIEKELMAN, P., AND F. GOULD. 2000a. Pest control by the introduction of a conditional lethal trait on multiple loci: potential, limitations, and optimal strategies. Journal of Economic Entomology 93:1543-1565.

SCHLIEKELMAN, P., AND F. GOULD. 2000b. Pest control by the release of insects carrying a female-killing allele on multiple loci. Journal of Economic Entomology 93:1566-1579.

SCHLIEKELMAN, P., S. ELLNER, AND F. GOULD. 2005. Pest control by genetic manipulation of sex ratio. Journal of Economic Entomology 98:18-34.

SCHUBERT, D., B. LECHTENBERG, A. FORSBACH, M. GILS, S. BAHADUR AND R. SCHMIDT. 2004. Silencing in *Arabidopsis* T-DNA transformants: the predominant role of a gene-specific RNA sensing mechanism versus position effects. Plant Cell 16:2561-2572.

SHARPE, R. M., S. M. MAGUIRE, P. T. K. SAUNDERS, M. R. RUSSELL, D. GANTEN, S. BACHMANN, L. MULLINS, AND J. J. MULLINS. 1995. Infertility in a transgenic rat due to impairment of cytoplasmic elimination and sperm release from the Sertoli cells. Biology of Reproduction 53:214-226.

SIBLEY, R. M., D. BARKER, M. C. DENHAM, J. HONE, AND M. PAGEL. 2005. On the regulation of populations of mammals, birds, fish and insects. Science 309:607-610.

STRONG, D. R., AND R. W. PEMBERTON. 2000. Biological control of invading species - risk and reform. Science 288:1969-1970.

SULLIVAN, R. 2004. *Rats*. Bloomsbury, New York, USA.

TESSON, L., J. COZZI, S. MENORET, S. REMY, C. USAL, A. FRAICHARD, AND I. ANEGON. 2005. Transgenic modifications of the rat genome. Transgenic Research 14:531-546.

THOMAS, D. D., C. A. DONNELLY, R. J. WOOD, AND L. S. ALPHEY. 2000. Insect population control using a dominant, repressible, lethal genetic system. Science 287:2474-2476.

THRESHER, R. E., AND N. BAX. 2006. Comparative analysis of genetic options for controlling invasive populations of the cane toad, *Bufo marinus*. Pages 117-122 *in* K. Molloy, and W. Henderson, editors. Science of cane toad invasion and control. Proceedings of the Invasive Animals CRC/CSIRO/Qld NRM&W Cane Toad Workshop, June 2006. Brisbane, Australia. Invasive Animals Cooperative Research Centre, Canberra, Australia.

THRESHER, R. E., AND A. KURIS. 2004. Options for managing invasive marine species. Biological Invasions 6:295-300.

THRESHER, R. E., P. GREWE, J. PATIL, AND L. HINDS. 2005. Genetic control of sex ratio in animal populations. Australian Patent Number 2001291520, Commonwealth Scientific and Industrial Research Organisation.

TORRES, J. M., C. SANCHEZ, M.A. RAMIEREZ, M. MORALES, J. BARCENA, J. FERRER, E. ESPUNA, A. PAGES-MANTE AND J. M. SANCHEZ-VIZCAINO. 2001. First field trial of a transmissible recominbinant vaccine against myxomatosis and rabbit hemorrhagic disease. Vaccine 19:4536-4543.

TYLER, M. J. 1998. Australian frogs. A natural history. Cornell University, Ithaca, New York, USA.

VEITCH, C. R., AND M. N. CLOUT. 2002. Turning the tide: the eradication of island invasive species. Auckland, Invasive Species Specialist Group of the The World Conservation Union (IUCN).

WHITTEN, M. J., AND G. G. FOSTER. 1975. Genetical methods of pest control. Annual Review of Entomology 20:461-476.

WIBBELS, T., AND D. CREWS. 1994. Putative aromatase inhibitor induces male sex determination in a female unisexual lizard and in a turtle with temperature-dependent sex determination. General Comparative Endocrinology 141:295-299.

WILHELM, D., AND P. KOOPMAN. 2006. The makings of maleness: towards an integrated view of male sexual development. Nature Reviews Genetics 7:620-631.

# ALASKA'S RAT SPILL RESPONSE PROGRAM

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***Abstract:*** The introduction of rodents on an island as a new predator usually interferes with natural island biodiversity, particularly on islands without any native mammalian predators. Many Alaskan islands, and most islands in the Aleutian Island region of the Alaska Maritime National Wildlife Refuge (AMNWR), are free of mammalian predators and are vulnerable to invasion by rodents. Rat introduction to islands can put ground-nesting birds, such as seabirds and endemic landbirds, at risk of extirpation. The refuge is also concerned about additional introductions of house mice. As far as we know, the introduction, or "spilling," of rats onto refuge islands from ships and cargo was accidental, but probably preventable. This paper is about preventing new rodent invasions, especially rats, on Alaskan islands from shipwrecks, and using our experience on AMNWR as a basis for recommendations about improvements in the future.

***Key Words:*** Alaska islands, invasive species, rapid response, rodents, rodenticide, seabirds, ship, shipwreck.

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## INTRODUCTION

Rodents, such as Norway rats (*Rattus norvegicus*), have successfully invaded many areas outside their native range and became established throughout the world. In Alaska, rats are established on least three mainland communities and three communities on Southeast Alaskan islands. Additionally rats breed on some islands in the Aleutian chain of islands. Most Aleutian Islands are part of the Alaska Maritime National Wildlife Refuge (AMNWR), which includes more than 2500 islands and 3.5 million acres. Currently, Norway rats are established on 11 large refuge islands and numerous associated smaller islets (Bailey 1993). Black rats (*Rattus rattus*) are known to occur on one Aleutian Island (Taylor and Brooks 1995).

There several pathways of invasion for rats to Alaskan islands. Rats can reach islands by climbing or swimming from ships, riding ashore in cargo, or escaping from shipwrecks. Rats can eventually reach nearby islets from infested islands by riding floating debris, crossing kelp bridges between islands, or swimming.

Rats invade ships in foreign harbors, United States (US) harbors, and some Alaskan ports. A few islands with refuge land have harbors (Shemya, Adak, Atka, Unalaska, Akutan, Popof, St Paul, St George) with facilities where ships can tie up, but of these, only the islands of Shemya, Adak, Atka, Unalaska and Akutan currently have breeding populations of rats. Additionally, off-refuge Alaskan commercial harbors of Juneau, Ketchikan, Nome, Petersburg, and Sitka have breeding populations of Norway rats.

The Aleutian Islands are on the great circle shipping route, a favorite route of international ships traveling between the US west coast and Asia. An estimated 31,000 ships use the route annually (NUKA Research & Planning Group, LLC 2006), and some might carry rodents that could infest islands if wrecked on Alaskan shores. Norway rats have invaded one Aleutian Island (Rat Island), because of a shipwreck (Masterson and Brower 1948), and possibly rats invaded other islands from shipwrecks. Prior to World War II (WWII), Alaskan islands were likely invaded by rats from ships in harbors such as Kodiak, Akutan, Unalaska, Attu, and Atka, during the wooden ship days.

During WWII, islands were occupied by Japanese, Canadian and US troops. Rats might have first invaded these military outposts from cargo transported by planes or ships, or from ships directly. Among other islands, airstrips were built on Attu, Shemya, Amchitka, and Adak, and piers were built on Attu, Kiska, and Great Sitkin, where rats became established at this time. Kagalaska might have been invaded after rats became established on Adak Island, as happened on dozens of smaller islets near these large islands. In spite of introductions, rats have not become established on all refuge islands, but reasons why are unclear. The

threat of more refuge islands becoming infested with rats will exist as long as ships harboring rats steer towards or drift on rat-free islands when they are disabled.

AMNWR started working with the communities of the Pribilof Islands in 1993 to keep rodents from becoming established on the islands of St. Paul and St. George. AMNWR later developed a strategy for combating potential rat invasions resulting from shipwrecks. The immediate goal was to prepare for response to a potential "rat spill" from disabled ships to adjacent shorelines. A long-term goal is to maintain a multi-agency team that is trained, experienced, and supplied to prevent new infestations of rodents on islands. Specifically, the strategy is to coordinate with the US Cost Guard (USCG) to keep potentially infested ships from going aground on islands vulnerable to rat invasion. Potential actions include towing ships away from refuge islands and allowing the burning or sinking of distressed vessels. Many ships do not break up quickly when going aground, so boarding grounded vessels to inspect and kill rodents might be possible. Shorelines can be protected from escaping rats using traps and rodenticide bait.

One challenge with shipwreck events is that they occur suddenly, and first response time is critical to successfully preventing island invasions. It is safe to assume the number of rodents reaching shore increases with time after a shipwreck. The scale of the response necessary to capture fleeing rats can quickly increase beyond the availability of personnel or resources.

The primary concern about shipwrecks is for protecting human safety and minimizing environmental damage from an oil spill. However, AMNWR has elevated the concern for potential rodent invasion from shipwrecks with the USCG and others. Response to the potential of rodents escaping from shipwrecks now has been integrated with AMNWR's participation when disabled ships threaten refuge islands resources from oil spills. Rodent response training is concurrent with the larger, more formal oil spill response training.

## REFUGE RAT SPILL RESPONSE

The USCG has primary jurisdiction of management of disabled ships, regardless of location. Through the Coast Guard, the US Fish and Wildlife Service (USFWS) is notified of potential shipwrecks in Alaskan waters and near refuge islands.

The USFWS Alaska Region Oil Spill Coordinator, in Anchorage, is notified by the USCG when a ship in distress is likely to result in a shipwreck. If the ship is near refuge lands, or if it might travel on its own, be towed, or drift near refuge islands, the Coordinator notifies refuge staff. Using information available and first-hand knowledge of the resources at stake, refuge staff begin to evaluate the potential hazard of a rat invasion to refuge lands. Staff provide comments on proposed actions and concerns to the Coordinator, who is the refuge spokesperson to the USCG Incident Commander.

The refuge identifies the location of rat response supplies, notifies qualified staff able to respond to a potential rat spill, and plans logistics to get responders and supplies near the ship grounding. Initial rat spill kits are staged in several locations in Alaska (Adak, St. Paul, St. George, Unalaska, Homer, Anchorage and Juneau), including one first-response salvage ship. Additionally, the Alaska Department of Fish and Game maintains a rat spill response kit on Round Island State Wildlife Refuge. The refuge rat spill response strategy is compatible with a new State Rat Management Plan (draft in review).

Mobile spike camp kits are located in Adak, Homer, and Anchorage. Spike camp kits include basic camping and survival gear for two people to stay on-scene in remote Alaskan locations. The kits are designed as a starting point for outfitting a rat response headquarters. It does not preclude the necessity of transporting additional supplies, such as food and fuel, to the incident scene.

When a ship grounding on the refuge is imminent, refuge staff are deployed to the Incident Command Center, and refuge headquarter staff are dedicated to their support. On-the-scene assessment of the disabled ship and potential invasion points is critical to evaluating the risk of a potential rat spill. Access to the disabled ship often is difficult because of the hazardous environment, and priority always is given to human safety and a potential oil spill. Initial on-the-ground assessment of a potential rodent invasion usually occurs after ship grounding and later in the sequence of oil spill response.

The rat spill response issue is a concern with more than just islands within the Alaska Maritime National Wildlife Refuge. Worldwide, other islands have been invaded by rats from shipwrecks. The refuge recognizes opportunities to improve its Rat Spill Response strategy by examining the issue in a broader, world-wide context.

Near-misses of refuge islands are more common than actual groundings, but consultation between the USCG and the USFWS can reduce risk of groundings on sensitive islands by considering the impact of potential rat spills when evaluating response alternatives.

It is difficult for refuge staff to gain first-hand experience with implementing on-the-ground rat spill response. Since 1994, the refuge has responded on-the-ground to potential rat spills three times (on St. Paul Island in 1987, Chirikof Island in 2002, and Afognak Island in 2003). Since the refuge's rat spill program began, no ship known to have rats has wrecked on a rat-free refuge island.

However, logistic planning for response to potential rat spills has occurred numerous times. In 1997, the freighter Kuroshima ran aground on Unalaska Island, which already was infested with Norway rats. In 2004, the 738-foot Selendang Ayu, carrying soybeans, went aground and split apart on the same island. During July 2006, a 654-foot car carrier, the Cougar Ace, rolled onto its beam south of Adak Island and had to be towed to Dutch Harbor. In December 2006, another bulk grain carrier, the 534-foot Sea Honesty, developed engine problems in the Aleutians and was guided into Dutch Harbor for repairs. Each incident, regardless of how far the rat spill response progressed, provided a valuable experience for the people involved.

## RAT RESPONSE REGULATORY ISSUES

### Personnel Training and Certification

AMNWR is one of more than five hundred federal wildlife refuges administered by the USFWS. Relative to the size of the refuge, the refuge staff is small. Even if the entire permanent biological staff were trained, certified, and available during a shipwreck, the force would be small compared with the manpower used on the smallest successful island rat eradication. To help maintain an adequate pool of certified pesticide applicators, AMNWR, in cooperation with Alaska Department of Environmental Conservation (ADEC), has sponsored pesticide applicator training that leads to certification. The intent of this training is to qualify more than AMNWR field staff for the application of anticoagulant rodenticides if necessary during a shipwreck event.

### NEPA and FIFRA

The AMNWR has completed an Environmental Assessment (USFWS 1993) that satisfies the National Environmental Policy Act. The EA includes federal action to respond to rat spills on refuge islands. Additionally, a strategic plan to protect island ecosystems in Alaska from the introduction of rodents, including shipwreck response, was prepared in 1995 (DeGange et. al 1995).

According to the Federal Insecticide, Fungicide and Rodenticide Act (FIFRA), pesticide products must be approved and registered by the US Environmental Protection Agency (EPA), and must be used according to the label approved by EPA. Section 18 of FIFRA authorizes EPA to allow states to use a pesticide for an unregistered use for a limited time if EPA determines that emergency conditions exist. Authorizations (commonly called emergency exemptions) are granted to state and federal agencies for a specific length of time to control emergency situations. In the past, AMNWR has satisfied FIFFA requirements under Sec. 18, Emergency Exemptions, for using pesticides during rat spills. Currently the refuge is in the process of renewing its Sec. 18 permit.

Pesticides products selected for use in rat spill responses must be approved by the Alaska Department of Environmental Conservation (ADEC). Pesticide regulations are not limited to their application, but also include requirements for transport, storage, disposal, and protection of workers.

In 2003, at the request of the refuge, HACO, Inc., (Madison, WI) manufacturer of Ramik Green (0.005% diphacimone) obtained a revised label (EPA Reg. No. 2393-498) under Section 24(c) of FIFRA for the use of Ramik Green to control Norway and black rats for wildlife conservation purposes on AMNWR. The label has provisions for using Ramik Green in bait stations and inside rodent burrows, both useful methods for preventing rodent invasions of islands from shipwrecks.

Recently a Federal Section 3 label was approved that is applicable to combat non-native rodents in Alaska shipwreck responses in Alaska: Diphacinone 50 Conservation (56228-35). According to this label, the anti-coagulant diphacinone bait may be applied on shore by aerial or hand-broadcast on the ground or in vegetation canopy, inside rodent burrows, and inside bait stations. On disabled vessels, baits may be placed in bait stations.

A Federal Section 3 label for another anti-coagulant bait, Brodifacoum 25 Conservation, is under consideration by EPA. The purpose of this registration is for conservation uses similar to the

Diphacinone 50 Conservation label. If the Brodifacoum 50 Conservation label is approved, it would be an important tool for shipwreck response, especially when invading rats may be resistant to first-generation rodenticides (such as diphacinone). Proposed application methods of Brodifacoum 25 Conservation, a second-generation anticoagulant, includes aerial and hand-broadcast, inside rodent burrows, and bait stations.

The US Department of Agriculture Wildlife Services National Wildlife Research Center and USFWS Pacific Islands Ecological Services coordinated these new agency registrations of rodenticides for control and eradication. The refuge supports these registrations for shipwreck, prevention, and eradication of invasive rodents and the re-registration of rodenticides useful for refuge applications.

Other formulations of rodenticides in Alaska are restricted to use around buildings and structures, which limits their applicability for shipwreck responses. However, these baits are useful for defending harbors and ships from rat infestation, which helps lower the risk of rat spills. EPA is evaluating the ecological risks with nine rodenticides, some of which are used in Alaska. EPA has proposed revising three of the nine rodenticide labels to make them restricted-use pesticides. Restricted-use pesticides can be applied only by certified users. Currently, these rodenticides can be purchased at home supply stores and other commercial outlets in Alaska and used according to the label directions by someone without a Pesticide Applicator's License. EPA also proposes requiring these rodenticides (and perhaps others) to be sold only in tamper-resistant bait stations. The ready availability of these rodenticides in urban stores reduces the need for harbor masters and boat owners to stockpile large amounts of pesticide in case of invasion of their facilities by rodents.

**Pesticide Use Proposal**

USFWS policy requires a Pesticide Use Proposal (PUP) before any federal action involving the application of pesticides. The refuge requests a PUP annually in case it needs to apply a rodenticide.

**Endangered Species**

The Endangered Species Act prohibits actions that have a negative impact on endangered species and their habitat. Under this statute, the USFWS must ensure that its pesticide use does not harm, threaten, or endanger species or their habitats. Endangered species that occur on the refuge include Steller's sea lions (*Eumetopias jubatus*), sea otters (*Enhydra lutris*), and Steller's eiders (*Polysticta stelleri*). The refuge considers all potential shipwreck strategies (choice of toxicant, bait application method, traps, and bait and device placement) to prevent negative impact on endangered species.

**Alaska Board of Game and Alaska Department of Fish and Game**

Currently, the USFWS has permission from the Alaska Board of Game to use rodenticides to take non-native rodents on refuge lands. Subsequently, the Alaska Department of Fish and Game authorized a Wildlife Nuisance Permit so refuge employees may take non-native rodents on islands within AMNWR. Under Alaska regulations, both authorities are necessary to use pesticides to kill wildlife, including rats, within the State.

**Other State Requirements**

Ultimately, states have primary responsibility for pesticides used within state borders. Individuals applying restricted-use pesticides must be certified pesticide applicators by ADEC. Certification is also a condition of the Sec. 18 Exemption granted to the refuge to apply baits in a shipwreck response.

A permit from ADEC is required to aerially apply pesticides by helicopter or aircraft within the State of Alaska. In the past, none of the pesticides proposed for use in shipwreck response has been registered in Alaska for aerial application. However, the recent label approved by EPA for Diphacinone 50 Conservation allows aerial broadcast applications. Aerial application likely is not feasible during or soon after a shipwreck because of the planning and resources necessary for an aerial operation. However, if aerial application is practical, the refuge may, at that time, pursue a permit through the ADEC.

Unlike applying pesticides, shooting or trapping rats does not require ADEC certification. However, a state hunting or trapping license is required to shoot or trap wildlife, including rats, in Alaska. However, refuge employees are is allowed to take non-native rodents on refuge lands without a sporting license under the authority of the Wildlife Nuisance Permit.

## RECOMMENDATIONS

### Federally-Coordinated Rapid Response Team

Increased roles for state and federal agencies are needed to facilitate and coordinate the response of federal or international rapid response teams. Federal coordination is needed for a Rapid Response Team for rat spills in the US, similar to rapid response teams called for by the National Aquatic Invasive Species Act. Team members would participate in training internationally, and provide training to state, local, tribal rapid responders. A multi-agency Alaskan team would specialize in rat-spill response in Alaskan waters. For shipwrecks potentially involving refuge lands or trust resources, response would initially be coordinated in Anchorage by the USFWS, and the refuge would be engaged as first-responders.

### Training

A workshop was conducted in 2004 to evaluate the refuge's invasive rodent program, and shipwreck response training was included. During that meeting, and subsequent refuge-sponsored and multi-agency rat spill workshops, participants worked through a rat spill scenario. These sessions always provided insights about how to best respond to shipwrecks and prevent rat spills. The refuge continues to provide rat spill response training. Rat response drills, modeled after oil spill response drills, would provide additional practice at integrating different aspects of shipwreck response.

### Building Rat Spill Response Capacity with Island Eradication

Rat invasions from shipwrecks could be treated as small-scale island eradications. If there is a predictable limit to range expansion after invasion, it may be possible to treat only a portion of large islands soon after invasion. The restoration of a refuge island in the Aleutians by eradicating Norway rats is a goal of AMNWR and its partners. This project will help AMNWR gain first-hand experience using rodenticide on a larger scale than its harbor prevention program.

The island rat eradication project currently proposed by AMNWR would rely upon the aerial broadcast of brodifacoum rodenticide pellets. Aerial dispersal of rodenticide bait is currently not approved in Alaska, and it is not the first approach that would be considered in shipwreck response. Aerial broadcast is a practical method to treat large areas for the protection of islands soon after shipwrecks, and it may become an important option if other prevention techniques fail.

In cooperation with The Nature Conservancy and Island Conservation, the refuge is in the planning stage for an eradication of rats from a 2,779 ha (6,861 ac) island by aerial broadcast, potentially as early as late summer 2008, and it would be the first aerial broadcast of rodenticide in Alaska. If successful, the project will demonstrate this method can be applied within the constraints of weather, non-target concerns, and the logistic challenges of Alaska.

Some issues are similar between an eradication of rats on islands after they are established, and prevention of rodents escaping from a shipwreck from becoming established on refuge islands. For example, disturbance to endangered species such as sea lions, sea otters, and Steller's eiders must be minimized, and these animals must be prevented access to areas treated for rats. Impacts of control operations on non-target native wildlife must be considered. As the refuge works through these issues for island rat eradication, we prepare ourselves to better resolve the same obstacles during a large-scale rat response.

If the restoration of Rat Island to rat-free status is successful, then other, larger islands in the Aleutians might be restored to productive sea bird habitat using similar methods if funding becomes available.

## LITERATURE CITED


BAILEY, E.P. 1993. Fox introductions on Alaskan islands - history, impacts on avifauna, and eradication. Washington, D.C.: U.S. Dept. of the Interior, Fish and Wildlife Service; Springfield, VA: National Technical Information Service, distributor.

DEGANGE, A. R., A. SOWLS, AND L. FAIRCHILD. 1995. A strategic plan to protect island ecosystems in Alaska from the introduction of rodents. Unpublished USFWS Report, Anchorage, Alaska, USA.

MASTERSON, J. R., AND H. BROWER. 1948. Bering's successors, 1745-1780, contributions of Peter Simon Pallas to the history of Russian exploration toward Alaska. University of Washington Press, Seattle, Washington, USA.

NUKA RESEARCH AND PLANNING GROUP, LLC. 2006 Vessel traffic in the Aleutians subarea. Prepared for the Alaska Department of Environmental Conservation, Contract 18-8003-28-09. NUKA, Seldovia, Alaska, USA.

TAYLOR, R. H., AND J. E. BROOKS. 1995. A survey of shemya rodents. Project Report. U.S. Fish and Wildlife Service Legacy Project 1244. Unpublished report, Alaska Maritime National Wildlife Refuge, Homer, Alaska, USA.

U.S. Fish and Wildlife Service. 1993. Environmental assessment proposed emergency use of toxicants to prevent accidental introductions of rats from shipwrecks on islands in the Alaska Maritime National Wildlife Refuge. Unpublished. Alaska Maritime National Wildlife Refuge, Homer, Alaska, USA.

# POPULATION ECOLOGY AND SOME POTENTIAL IMPACTS OF EMERGING POPULATIONS OF EXOTIC PARROTS

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***Abstract:*** Parrots have been transported around the world since the 1400s and about 44 species have established populations outside of their native ranges. At least 70% of these introduced species were identified as potential agricultural pests, compared to only 15% of all parrot species. At least 74 exotic psittacines have been reported in the wild in the United States, but only nine species were considered established. Since 1990, 24 psittacine species have been sighted in the wild in Hawai`i, about 21 species regularly occur on the main islands and at least five appear established. Past history of successful introduction, recent status of introduced populations, and life history traits were used to identify parrots which, if introduced, appeared most likely to establish nonindigenous populations. Potential impacts related to ecology, agriculture, and health were diverse but poorly documented in the literature. Five species were the most widespread and abundant species, and were considered established. Four more species with rapidly expanding ranges and increasing populations may become naturalized. Nine species had significantly expanded ranges and increased populations. Several introduced populations remained low for 10-15 years before rapidly increasing and becoming established; similar lag periods for other psittacines have approached 30 years.

***Key Words:*** Hawaii, invasive species, parrots, psittacines.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Exotic species invasions are a leading cause of native species declines and are a major environmental issue of public concern (Didhama et al. 2005). The establishment of introduced species is recognized as a major ecological threat worldwide, and increasing volumes of international trade are expected to increase the rate at which species are unintentionally transported (Ricciardi et al. 2000). Geographic and taxonomic patterns of exotic bird introductions are striking. In their analyses of 1,378 introductions of 426 bird species around the world, Blackburn and Duncan (2001a) found that 40% occurred in four geographic areas: Hawai'i (163 introduction events), New Zealand (146), mainland United States (US, 125) and Australia (113). On the 6 main Hawaiian Islands, at least 140 species of birds, belonging to 14 different taxonomic orders and originating from 6 continents have been introduced, including 75 species of gamebirds (Long 1981, Moulton et al. 2001). Taxonomic patterns of successfully established exotic birds are also distinctly nonrandom and 7 families contained more successfully established exotic species than expected by chance: Anatidae (15 species of ducks and geese), Phasianidae (35 species of pheasants), Passeridae (30 species of sparrows and finches), Psittacidae (39 species of parrots), Columbidae (19 species of pigeons and doves), Rheidae (2 species of rhea), and Odontophoridae (6 species of quail) (Lockwood 1999, Cassey 2002).

Loope et al. (2001) noted that in contrast to mainland states like Florida, inadvertent colonization of Pacific islands is highly unlikely and all successfully established exotic birds in Hawai`i were deliberately brought to the islands for some purpose. Limiting introductions therefore has relied mainly on regulating the trade in live birds and they predict that future introductions are likely to come from four groups: waterfowl, gallinaceous birds, psittacines, and passerines (especially finches). In line with this, Foster (In Press) documented that overall patterns of bird introductions into Hawai`i have shifted over time from intentional releases to escapes of birds from captivity; since 1976 this has included at least 9 psittacines.

In this paper, we highlight emerging populations of exotic parrots and review potential effects related to ecology, agriculture, and health. Few data are available from Hawai'i and we review information from naturalized parrot populations on the mainland US to identify species with increasing and expanding naturalized breeding populations, those with traits similar to successfully established psittacines (Cassey et al. 2004b) and with the potential to become agricultural pests if established.

Williamson (1996) identified 4 steps in the process of invasion of a region by exotic species as a result of human activity, and these steps have served as a basis for several subsequent studies of patterns of success for introduced birds. Blackburn and Duncan (2001a) labeled the stages of this process as "transport, introduction, establishment and spread." Usage of some relevant terms was not uniform and we followed the generalized definitions of Kolar and Lodge (2001). Indigenous species: a species found within its native range. Invasive species: a nonindigenous species that spreads from the point of introduction and becomes abundant. Nonindigenous species: a species introduced to areas beyond its native range by human activity; exotic species is commonly used synonymously. Noninvasive species are nonindigenous species that remain localized within a new environment. Established species are species with self-sustaining populations outside of the native range. Introduced birds are deemed established by the American Ornithologists Union (AOU 1998) "if there are persistent records for at least 10 years and satisfactory evidence of maintaining a reasonably stable or increasing population through successful reproduction." (i.e., not supported by continued releases). Long lifespan makes determining establishment in parrots more difficult than in other groups, but 30 species have been successfully introduced in at least one location outside of their native ranges, and introductions of an additional 14 parrot species were probably successful (Long 1981, Cassey et al. 2004a, b, Table 1).

## OVERVIEW OF PSITTACIFORMES

The Order Psittaciformes is a distinct group of birds with large hooked bills and zygodactyl feet. There are 2 families: Cacatuidae (cockatoos) and Psittacidae (parrots). Origins have been traced back to the Miocene in Queensland and Australia is considered the center of evolution and radiation for psittacines (del Hoyo et al. 1997). Diversity of parrots is highest in Australasia and South America with about one-third of all parrot species occurring in each regions. Comprehensive species accounts were compiled by Juniper and Parr (1998), Forshaw (1977), and del Hoyo et al. (1997); and online at www.arndt-verlag.com. This overview follows the taxonomic nomenclature of del Hoyo et al. (1997).

## INFLUENCES ON ESTABLISHMENT SUCCESS

Once established, the control or eradication of invasive species is highly costly, and preventing introduction and establishment is preferable (Kolar and Lodge 2001). This has prompted researchers to characterize established species in order to identify potentially invasive species before the invasion has occurred. Traits associated with invasive birds are as varied as migratory habits (Kolar and Lodge 2001), broad diet and habitat niches (Cassey et al. 2004b, c), and behavioral flexibility and forebrain size (Sol and Lefebvre 2000). Although there are intrinsic differences in the ability of bird species to invade new locations, various studies (e.g., Lockwood 1999, Sol and Lefebvre 2000) have confirmed that introduction effort, defined as the number of releases or numbers released at a location, is the principal factor influencing numbers of nonindigenous bird species.

Introduction effort is correlated with many variables previously thought to influence establishment success (Blackburn and Duncan 2001a, b), and highlights its importance both within and across large regions including oceanic islands (Cassey et al. 2005). Moulton et al. (2001) found a significant relationship between introduction effort and success rates of Galliformes (game birds) introduced to the Island of Hawai'i. Even though successful establishment ultimately depends on the particular combination of a species and a location (Blackburn and Duncan 2001a, b) this consistent finding regarding introduction effort suggests that fewer species will become established if the number of releases or number of individuals is minimized.

The Wild Bird Conservation Act of 1992 specifies that wild-caught birds may be imported into the US only if they come from a country with a US Fish and Wildlife Service (FWS) approved management plan for sustainable use of the species. Before this, the US was the world's largest

**Table 1.** Life history scores for Psittacines with successful and probably successful introduced populations.

| Species | Altitude range | Migration score | Diet breadth | Ag. pest? |
|---|---|---|---|---|
| Introductions successful: | | | | |
| Sulphur-crested cockatoo (*Cacatua galerita*) | 2,400 | 0 | 4 | y |
| Rainbow lorikeet (*Trichoglossus haematodus*) | 2,440 | 0 (1) | 3 | y |
| Kuhl's lorikeet (*Vini kuhlii*) | -- | 0 | 2 | n |
| Blue lorikeet (*Vini peruviana*) | -- | 1 | 4 | y |
| Ultramarine lorikeet (*Vini ultramarina*) | 1,000 | 0 | 3 | y |
| Red Shining-parrot (*Prosopeia tabuensis*) | 1,650 | 0 | 3 | y |
| Yellow-fronted parakeet (*Cyanoramphus auriceps*) | -- | 0 | 5 | n |
| Crimson rosella (*Platycercus elegans*) | 1,900 | 0 (1) | 4 | y |
| Eastern rosella (*Platycercus eximius*) | -- | 0 | 5 | y |
| Budgerigar (*Melopsittacus undulatus*) | -- | 2 (1) | 1 | y |
| Eclectus parrot (Eclectus roratus) | 1,900 | 0 | 4 | y |
| Rose-ringed parakeet (*Psittacula krameri*) | 2,000 | 0 | 2 | y |
| Red-breasted parakeet (*Psittacula alexandri*) | 2,000 | 0 (1) | 3 | y |
| Grey-headed lovebird (*Agapornis canus*) | 1,500 | 0 | 2 | y |
| Fischer's lovebird (*Agapornis fischeri*) | 900 | 0 | 2 | n |
| Meyer's parrot (*Poicephalus meyeri*) | 2,200 | 0 (1) | 4 | y |
| Blue-crowned parakeet (*Aratinga acuticaudata*) | 2,650 | 1 | 2 | y |
| Green parakeet (*Aratinga holochlora*) | 2,300 | 0 | 2 | y |
| Mitred parakeet (*Aratinga mitrata*) | 2,400 | 1 (0) | 2 | y |
| Red-masked parakeet (*Aratinga erythrogenys*) | 2,500 | 1 | 3 | n |
| Brown-throated parakeet (*Aratinga pertinax*) | 1,200 | 2 | 3 | y |
| Black-hooded parakeet (*Nandayus nanday*) | 800 | 1 | 4 | y |
| Monk parakeet (*Myiopsitta monachus*) | 3,000 | 0 (1) | 4 | y |
| Green-rumped parrotlet (*Forpus passerinus*) | 1,800 | 2 | 4 | n |
| White-winged parakeet (*Brotogeris versicolurus*) | 2,700 | 0 | 3 | n |
| Yellow-chevroned parakeet (*Brotogeris chiriri*) | 1,560 | 1 | 3 | n |
| Hispaniolan parrot (*Amazona ventralis*) | 1,500 | 0 | 2 | y |
| Red-crowned parrot (*Amazona viridigenalis*) | 1,200 | 1 | 4 | n |
| Lilac-crowned parrot (*Amazona finschi*) | 2,000 | 2 | 2 | y |
| Yellow-headed parrot (*Amazona oratrix*) | 500 | 0 (1) | 3 | y |
| Introductions probably successful: | | | | |
| Galah (*Eolophus roseicapillus*) | 1,250 | 1 | 4 | y |
| Long-billed corella (*Cacatua tenuirostris*) | -- | 0 | 4 | y |
| Little corella (*Cacatua sanguinea*) | -- | 1 | 5 | y |
| Yellow-crested cockatoo (*Cacatua sulphurea*) | 1,200 | 0 | 4 | y |
| Blue-streaked lory (*Eos reticulata*) | -- | 0? | -- | n |
| Red-rumped parrot (*Psephotus haematonotus*) | 1,100 | 0 | 3 | n |
| Alexandrine parakeet (*Psittacula eupatria*) | 1,600 | 0 (1,2) | 4 | y |
| Plum-headed parakeet (*Psittacula cyanocephala*) | 1,500 | 0 | 4 | y |
| Rosy-faced lovebird (*Agapornis roseicollis*) | 1,500 | 0 | 2 | y |
| Yellow-collared Lovebird (*Agapornis personatus*) | 700 | 0 | 1 | n |
| Nyasa lovebird (*Agapornis lilianae*) | 1,000 | 0 | 3 | n |
| Chestnut-fronted macaw (*Ara severa*) | 1,150 | 0 (2)[4] | 2 | n |
| Yellow-crowned parrot (*Amazona ochrocephala*) | 750 | 0 (1) | 2 | y |
| Orange-winged parrot (*Amazona amazonica*) | 1,200 | 0? | 2 | y |

importer of parrots from wild populations. At least 150,000 live parrots were imported in 1990 (World Wildlife Fund 2007). Since passage of this Act, parrot imports have decreased, and captive breeding for the pet trade has increased. Since 1990, numbers imported have declined to about 17,000/year (World Wildlife Fund 2007). Mass importation, and thus mass releases and escapes, of wild-caught parrots into the US appear to be greatly reduced (Brightsmith 1999).

International restrictions in wildlife trade have similarly reduced parrot exports. From 1989-90 to 1999, numbers exported from the five main producer countries (Argentina, Guyana, Indonesia, Senegal, and Tanzania) dropped from 245,000 to 57,360 birds. Although actual origins are uncertain, 9 of the species recently sighted in Hawai`i were exported from these 5 countries. Exports of 8 of the species dropped from 88,790 to 26,290; in contrast, burrowing parakeets (*Cyanoliseus patagonus*) from Argentina entered the trade during this period and nearly 7,500 were exported (Roe et al. 2002). Despite increased regulation and quarantine requirements 2,811 parrots of 28 species were legally imported into Hawai`i in 1999 (Foster 2005, Hawai`i Department of Agriculture, unpublished data).

Future feral populations will most likely originate from small numbers of captive-bred pet parrots, rather than from large numbers of wild-caught birds. Thus, the process of establishment will likely be slow, requiring perhaps several decades, and the probability that additional parrot populations will become established should now be relatively low. However, releases of wild-caught birds being traded illegally are still possible. Cassey et al. (2004a, b) closely examined the patterns of transport, release and establishment of parrots around the world, and while the availability of individual birds for transport and release was very important, this had no obvious effect on establishment following release. Their multivariate statistical study revealed significant and independent effects of diet breadth, elevational range, and migratory habit. Sedentary, non-migratory, parrots and those with broad diets or large elevation ranges had higher establishment success. Other significant traits included lower fecundity, and longer fledgling periods.

At least 74 exotic psittacines have been reported in the wild in the US, but only 9 species are established (Table 2). Prior to 1981, 14 parrots had been introduced to the main Hawaiian islands

**Table 2.** Naturalized psittacines in the United States currently recognized by the American Ornithologists Union (AOU 1998, Banks et al. 2002), the California Bird Records Committee (2006), the Florida Ornithological Societies' Records Committee (2005), the Texas Ornithological Societies' Bird Records Committee (2006), Hawai`i Audubon Society (Pyle 2002) or the American Birding Association (ABA 2005).

| Species | AOU | California | Florida | Texas | Hawai'i | ABA |
|---|---|---|---|---|---|---|
| Budgerigar (*Melopsittacus undulatus*) | √ | | √ | | | √ |
| Rose-ringed parakeet (*Psittacula krameri*) | √ | | | | √ | |
| Mitred parakeet (*Aratinga mitrata*) | √ | | | | √ | |
| Green parakeet (*Aratinga holochlora*) | | | | √ | | √ |
| Black-hooded Parakeet (*Nandayus nenday*) | | | √ | | | |
| Monk parakeet (*Myiopsitta monachus*) | √ | | √ | √ | | √ |
| White-winged parakeet (*Brotogeris versicolurus*) | √ | | √ | | | √ |
| Yellow-chevroned Parakeet (*Brotogeris chiriri)* | | | | | | √ |
| Red-crowned parrot (*Amazona viridigenalis*) | √ | √ | | √ | √ | |

**Table 3.** Parrots introduced to Hawai`i, 1877-1981 (Moulton et al. 2001, Berger 1981, and Long 1981).

| Species | Islands | Years |
|---|---|---|
| Galah (*Eolophus roseicapillus*) | O`ahu | 1933 |
| Sulphur-crested cockatoo (*Cacatua galerita*) | O`ahu | 1933 |
| Salmon-crested cockatoo (*Cacatua moluccensis*) | O`ahu | 1981 |
| Bugerigar (*Melopsittacus undulatus*) | O`ahu | 1933 |
| Pale-headed rosella (*Platycercus adscitus*) | Maui | 1877 |
| Eclectus parrot (*Eclectus roratus*) | O`ahu | 1981 |
| Rose-ringed parakeet (*Psittacula krameri*) | Kau`ai, O`ahu, Hawai`i | 1981, 1933, 1981 |
| Peach-faced lovebird (*Agapornis roseicollis*) | O`ahu | 1973 |
| Scarlet macaw (*Ara macao*) | O`ahu | 1933 |
| Black-hooded parakeet (*Nandayus nenday*) | O`ahu, Hawai`i | 1981, 1981 |
| Monk parakeet (*Myiopsitta monachus*) | O`ahu | 1970 |
| Orange-chinned parakeet (*Brotogeris jugularis*) | O`ahu | 1933 |
| Yellow-crowned parrot (*Amazona ochrocephala*) | O`ahu | 1969 |
| Red-crowned parrot (*Amazona viridigenalis*) | O`ahu | 1971 |

(Berger 1981, Long 1981, Moulton et al. 2001, Table 3), but few are considered established (Pratt et al. 1987, Pyle 2002), at least one has been (Pyle 2002). Since 1970 (R. Pyle, unpublished data), 28 psittacines have been sighted in the wild in the state, species richness was highest on the islands of O`ahu (21) and Hawai`i (13). Currently, about 21 species regularly occur on the main islands including 11 breeding species (Table 4), and a few more species now appear to be established. Some species are being controlled and, hence, may not become established.

## ESTABLISHMENT PATTERNS

Generally, only a small proportion of exotic birds introduced to new locations have established wild populations, and have created negative impacts in the new environment (Lockwood 1999). Overall, less than 3% of all avian species have established nonindigenous populations somewhere in the world (Lockwood 1999) and 36% of alien birds introduced to Hawai'i have established breeding populations (Loope et al. 2001). Overall, parrots had even higher rates of establishment: 70% of 54 species intentionally released established naturalized populations (Cassey et al. 2004b). For long-lived birds like parrots, it can be hard to know if they are truly established, or are simply living out a long life span in a new location.

The highest diversity and abundance of nonindigenous parrots in the US occurs primarily in the subtropical areas of Florida, California and Texas (Pranty and Garrett 2003). The proportion of introduced and naturalized species is greater in urban/suburban areas than in rural areas and parrots fit this pattern (Spreyer and Bucher 1998, Lamont 2002, Butler 2003, Domenech et al. 2003, Eguchi and Amano 2004). Additionally, parrots usually exploit habitats dominated by exotic flora (Lamont 2002). In a few cases, however, a parrot species will colonize important agricultural areas (Domenech et al. 2003).

The occurrence of feral parrots non-urban/suburban areas may be due to a relatively large pool of potential escapees. Compared to rural areas, densities of pet parrots, like people, are likely much higher in urbanized areas; both density and diversity of feral parrots is especially high near ports of entry (e.g., Miami, Honolulu). Like southern California and southeastern Florida, Hawai`i has a mild climate, urban areas extensively landscaped with exotic vegetation, and major ports of entry. This, along with a lack of natural predators and abundant sources of invasive exotic

**Table 4.** Parrots possibly breeding and reported from the wild in Hawai`i since 1990. Species in bold typeface were identified as agricultural pest species, or as feeding on crops, orchards etc., in their native ranges by del Hoyo et al. (1997).

| Species | Islands |
|---|---|
| 11 breeding species: | |
| **Tanimbar cockatoo** (*Cacatua goffini*) | Maui |
| **Sulphur-crested cockatoo** (*Cacatua galerita*) | Maui |
| **Rose-ringed parakeet** (*Psittacula krameri*) | Hawai`i, Kaua`i, Maui, O`ahu |
| **Peach-faced lovebird** (*Agapornis roseicollis*) | Maui |
| Blue and yellow macaw (*Ara ararauna*) | Maui |
| Military macaw (*Ara militaris*) | Maui |
| **Scarlet-fronted parakeet** (*Aratinga wagleri*) | Hawai`i |
| **Mitred parakeet** (*Aratinga mitrata* ) | Maui |
| Red-masked parakeet (*Aratinga erythrogenys*) | O`ahu |
| **Burrowing parakeet** (*Cyanoliseus patagonus*) | Hawai`i |
| Red-crowned parrot (*Amazona viridigenalis*) | O`ahu |
| 13 species sighted since 1990: | |
| Salmon-crested cockatoo (*Cacatua moluccensis*) | Hawai`i, Maui, O`ahu |
| **Cockatiel** (*Nymphicus hollandicus*) | All |
| Black lory (*Chalcopsitta atra*) | Hawai`i |
| **Budgerigar** (*Melopsittacus undulatus*) | Maui, O`ahu |
| **Eclectus parrot** (*Eclectus roratus*) | Hawai`i, Kaua`i, O`ahu |
| **Alexandrine parakeet** (*Psittacula eupatria*) | Maui |
| **African grey parrot** (*Psittacus erithacus*) | Hawai`i |
| **Senegal parrot** (*Poicephalus senegalus*) | Hawai`i, Kaua`i, O`ahu |
| **Brown parrot** (*Poicephalus meyeri*) | Kaua`i |
| **Blue-crowned parakeet** (*Aratinga acuticaudata*) | O`ahu |
| **Black-hooded parakeet** (*Nandayus nenday*) | Hawai`i |
| **Yellow-crowned parrot** (*Amazona ochrocephala*) | O`ahu |
| Blue-fronted parrot (*Amazona aestiva*) | Maui, O`ahu |

food plants, likely increases the probability that introduced parrots will establish naturalized populations of psittacines. Three of the 9 species established on the mainland US (Table 2) are present and established in Hawai`i and if introduced, the other 6 presumably have the capacity to become established.

## METHODS

We compiled and tabulated life history information (T. Pratt and J. Foster, unpublished data, Forshaw 1977, del Hoyo et al. 1997, Juniper and Parr 1998). Introduction histories were gleaned from Garret et al. (1997), Simberloff et al. (1997), Lever (1987), Long (1981), and reviews of official state lists of bird status and occurrence for California, Florida and Texas. Using these data we calculated scores for three of the traits shown by Cassey et al. (2004b) to be correlated with successful establishment of introduced psittacines: extent of altitudinal range, diet breadth, and degree of migration. The scores used in their analyses were not available to us, and we calculated them based on the descriptions in Cassey et al. (2004b). Altitude range was the difference between the normal upper and lower altitudinal range limits of a species in its natural range. Diet breadth reflected the composition of each species diet scored as presence or absence of five different food types: fruit/berry, seed/grain, vegetative material, animal

material or nectar/flower. Each diet category was scored as 0 or 1 and summed (range 1 to 5). Cassey et al. (2004b) scored diets based on the first four of these categories only. Degree of migration was scored as: (0) for resident species with local movements only, (1) for nomadic species with irruptive seasonal movements, and (2) for species with regular altitudinal or geographical migrations. Unlike the previous two variables, migration score was inversely related to establishment success (Cassey et al. 2004b). We scored migration habit for a limited number of parrots which scored highly for altitudinal range and diet breadth, or which we judged potentially important based on their likelihood of establishment success, presence in Hawai'i, etc. For a similarly limited number of species we also noted if they were identified as agricultural pests, or reportedly fed on crops or orchards in their native ranges (del Hoyo et al. 1997).

In an effort to identify psittacines that could become established, and perhaps invasive, we searched data from the National Audubon Society's (NAS) annual Christmas Bird Counts (CBC) and looked for parrots that were both widespread and abundant. Our figures were created from the data accessible from the Audubon website: www.audubon.org/bird/cbc. To examine relative distribution among species we used the number of count circles where a species was reported in each count year for each state. For abundance we used the sum of all individuals reported in each count year for each state. Because we were most interested in total numbers of birds reported, rather than in statistical trend analyses, we have presented these data uncorrected for survey effort, and have made no inferences to larger geographical areas outside of the CBC circles. However, we note that there are both advantages and limitations to using CBC data to examine status of exotic birds (Pranty 2002, Butler 2005).

To identify broadly distributed psittacines with large populations, we examined counts from seven recent CBC seasons (winters of 1999-2000 through 2005-2006) from the three states with the highest diversity of exotic parrots: California, Florida and Texas. As a relative index to the extent of distribution within each state we used the maximum number of count circles that reported each species during this time period. The maximum of the annual sum of all individuals reported was used to index each species abundance. To facilitate visual comparisons among species these two index values were used as x,y coordinates to plot distribution on the horizontal axis and abundance on the vertical axis.

We used similar metrics to portray recent changes in distribution and abundance with indices based on changes in numbers of CBC count circles and numbers of birds counted for each of the three states. For each species, changes in distribution were indexed as:

No. of count circles reported in 2006 ÷
No. of count circles reported in 2000

Changes in abundance were similarly indexed as:
Total no. of individuals reported in 2006
÷ Total no. reported in 2000

Our goal was to highlight species that had clearly shown the ability become both widespread and abundant. Our measures of distribution and abundance are ordinal indexes only, not statistical estimates. For inferences comparing abundances of different species and changes over time, statistical analyses were limited to nonparametric, rank order methods using SAS® version 9.1 (SAS Institute, Cary, NC). Spearman rank correlation coefficients ($R_s$) were used to analyze long-term (>10 years) trends in abundance and distribution of exotic parrot species on CBC circles within California and Florida; data from Texas were too sparse to analyze. To identify species with significant trends over time, $R_s$ was calculated by year with number of individual birds reported; and for year with number of CBC circles where a species was reported. $R_s$ values, and significance levels, reflected how closely the rank order of each variable corresponded with year. Inferences here were limited to concluding whether or not trends over time were statistically significant at the 10% level; we did not calculate trend lines or estimate rates of population growth.

## RESULTS AND DISCUSSION

Psittacines with successful, or probably successful, introduced populations somewhere in the world are listed in Table 1. Most were from the subfamily Psittacinae, but five cacatuids, and four loriinids were also included. An established species becomes invasive when both distribution and abundance increase, and data from CBC circles (NAS 2002) demonstrated that monk parakeets, from South America, have increased in both

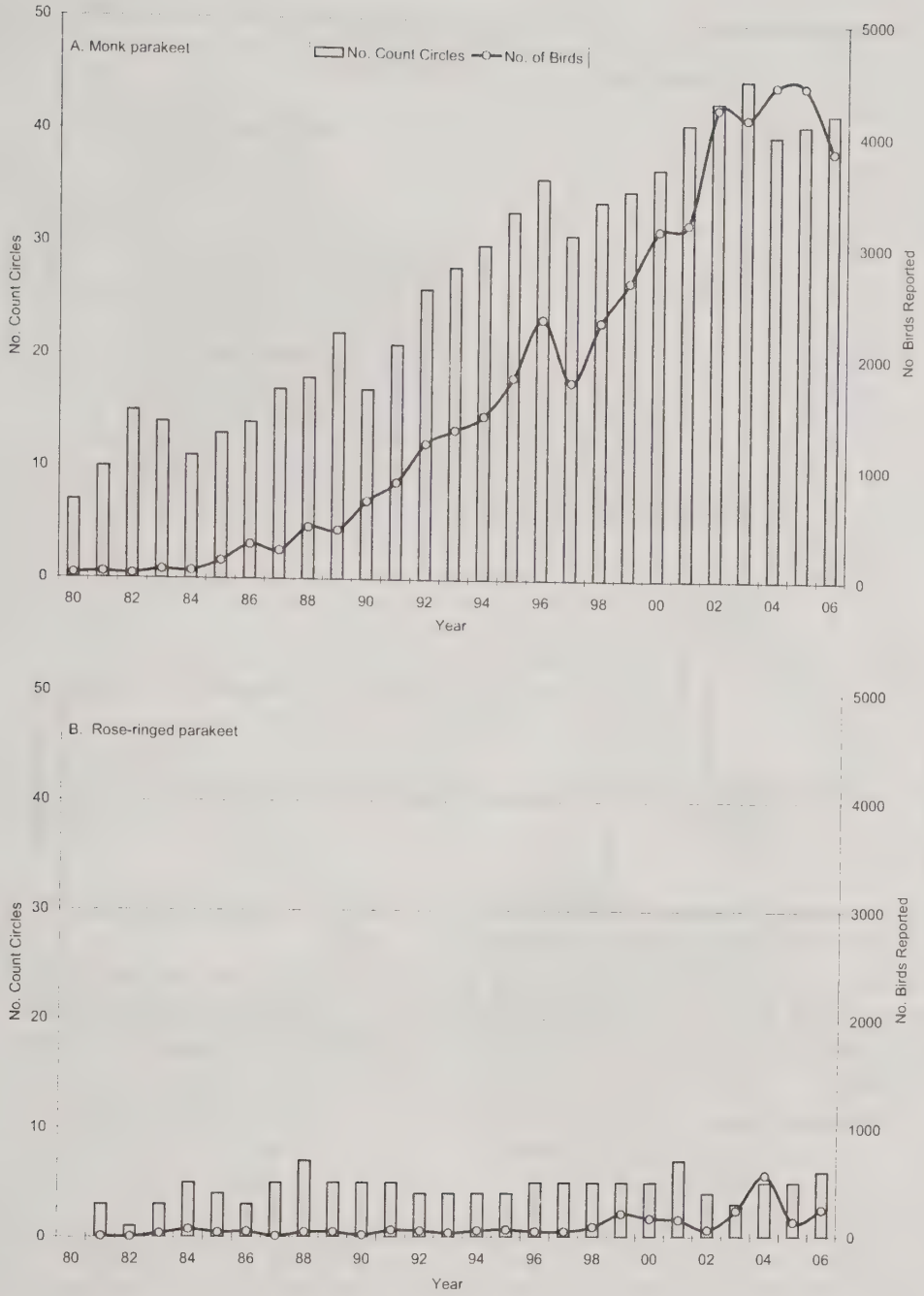



**Figure 1.** Trends in distribution and abundance of monk parakeets and rose-ringed parakeets from CBC circles within the US, 1980-2006.

numbers and distribution in North America (Figure 1a). Originating from multiple releases in the 1960s, they have recently bred in 11 states and Puerto Rico (Spreyer and Butcher 1998). At the same time, rose-ringed parakeet populations, introduced to the US in the mid-1960s to 1970s, have remained much more limited (Figure 1b) and were largely restricted to southern California, south Florida and Kaua'i. Patterns of population growth for established psittacines in Florida varied (Figures 2, 3) and included a period of slow growth followed by rapid expansion in both range and population size for monk (*Myiopsitta monachus*) and black-hooded parakeets (*Nandayus nenday*); both are now established members of the local avifauna. The initial high growth rate and spread of the budgerigar population, fueled by large scale releases of captive-bred birds, was followed by a sudden collapse (Figure 2b). Such crashes are common among naturalized species (Simberloff and Gibbons 2003) and exotic birds whose populations have exploded may not remain abundant. Examples from Hawai'i include the common myna (*Acridotheres tristis*), red-billed

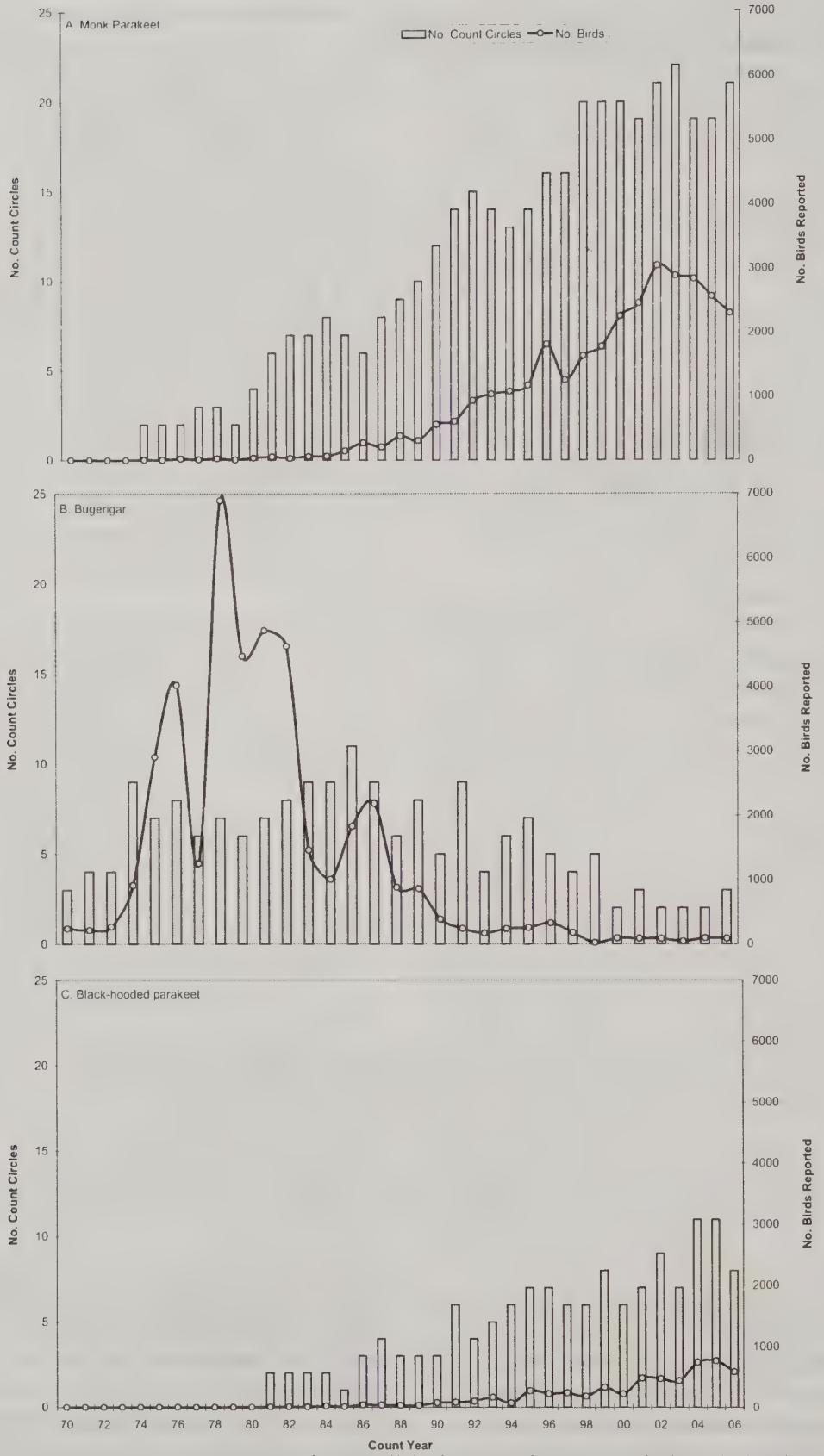



**Figure 2.** Trends in distribution and abundance of 3 established parrots from CBC circles within Florida, 1970-2006. Monk parakeets are invasive pests but may be declining (A); bugerigars range has contracted and numbers have plummeted (B); black-hooded parakeets are recently established (C).

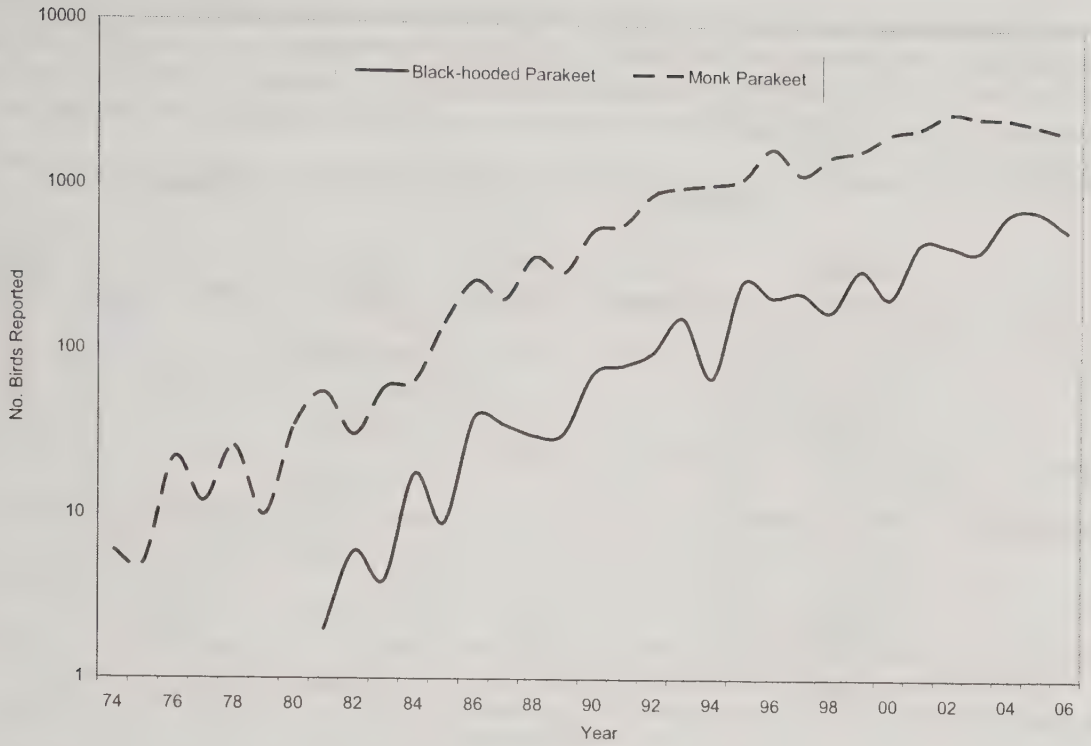



**Figure 3.** Trends in numbers of black hooded parakeets (solid line) and monk parakeets (dashed line) from CBC circles within Florida, 1974-2006.

leiothrix (*Leiothrix lutea*), and the varied tit (*Parus varius*, Foster 2005).

There were sufficient data to examine long-term (11 to 33 years) trends in distribution and abundance for 15 species in California and Florida (Table 5). Mitred (*Aratinga mitrata*), black-hooded and yellow-chevroned parakeets (*Brotogeris chiriri*) each had significant ($P \leq 0.10$) positive correlations over time for both numbers of individual birds reported and numbers of CBC circles within both states. Hence, both numbers and distribution increased over time. Conversely, budgerigars (*Melopsittacus undulatus*), yellow-headed parrots (*Amazona oratrix*), and red-crowned parrots (*Amazona viridigenalis*) showed significant negative correlations, indicating declines over time in Florida.

Data from CBC circles within California and Texas were adequate to portray the relative status of 16 species (Figures 4, 5), and 26 species in Florida (Figures 6, 7). These graphic summaries provided a snapshot of the relative status of different species on a state-by-state basis; widespread and abundant parrots appeared in the upper right, while smaller relatively localized populations appeared closer to the origin (Figures 4, 6). Populations which may have been relatively small and limited geographically, but which recently showed rapid increases in distribution and abundance appeared further to the right and above those with slowly expanding populations (Figures 5, 7). Numbers of three *Brotogeris* parakeets counted on CBC circles within California and Florida peaked at 200-300 (Figures 8, 9). Similarly for five species of *Aratinga* parakeets, numbers peaked at <100 to >300 (Figures 8, 10).

Rose-ringed parakeets (*Psittacula krameri*) were first recorded from CBC circles in Florida in 1974 (Figure 11), but were not reported from California until 10 years later with numbers relatively low for many years. Similarly, black-hooded parakeets were first recorded in Florida in 1981, about 10 years before they were reported from California, and were well established in south Florida by the early 2000s. Trends for five *Amazona* species are shown in Figures 12-14. Numbers of three *Amazona* species counted on CBC circles within California and Florida peaked at less than 50 birds, numbers of two others were higher (Figures 12-14).

Pranty (2001, 2002) used CBC data to graph population changes over time for 10 species of exotics, including three psittacines. Brightsmith (1999) used data from specific CBCs in south Florida and Los Angeles to help describe changes

**Table 5.** Long-term (>10 years) trends in abundance and distribution of exotic parrot species on CBC circles within California and Florida, 1970 to 2006. Values are Spearman rank correlation coefficients ($R_s$) for year with number of individual birds reported; and for year with number of CBC circles where a species was reported. Species with significantly increasing and expanding populations in both states are in bold typeface.

| | California | | | Florida | | |
|---|---|---|---|---|---|---|
| **Species** | **Years (no. yrs)** | **$R_s$ (abundance)** | **$R_s$ (CBC circles)** | **Years (no. yrs)** | **$R_s$ (abundance)** | **$R_s$ (CBC circles)** |
| Budgerigar | --- | --- | --- | 1970-2006 (30) | -0.55 *** | -0.22 |
| Rose-ringed parakeet | 1984-2006 (23) | 0.84 *** | 0.81 *** | 1974-2006 (33) | 0.22 | 0.32 |
| Blue-crowned parakeet | --- | --- | --- | 1989-2006 (18) | 0.61 *** | 0.42 * |
| Green parakeet | --- | --- | --- | 1984-2006 (23) | 0.39 * | 0.30 |
| **Mitred parakeet** | 1991-2006 (16) | 0.70 *** | 0.60 ** | 1989-2006 (18) | 0.61 *** | 0.45 * |
| **Red-masked parakeet** | 1994-2006 (13) | 0.76 *** | 0.73 *** | 1989-2006 (18) | 0.60 *** | 0.41 * |
| White-eyed parakeet | --- | --- | --- | 1989-2006 (18) | 0.58 ** | 0.45 * |
| Black-hooded parakeet | 1990-2006 (17) | 0.73 *** | 0.30 | 1981-2006 (26) | 0.97 *** | 0.94 *** |
| Monk parakeet | --- | --- | --- | 1974-2006 (33) | 0.99*** | 0.97 *** |
| **Yellow-chevroned parakeet** | 1996-2006 (11) | 0.67 ** | 0.90 *** | 1995-2006 (12) | 0.79 *** | 0.64 ** |
| Red-crowned parrot | 1976-2006 (31) | 0.86 *** | 0.86 *** | 1975-2006 (32) | -0.16 | -0.33 * |
| Lilac-crowned parrot | 1979-2006 (28) | 0.38 ** | 0.47 ** | --- | --- | --- |
| Blue-fronted parrot | --- | --- | --- | 1995-2006 (12) | 0.69 ** | 0.07 |
| Yellow-crowned parrot | 1975-2006 (32) | -0.25 | -0.05 | 1978-2006 (29) | -0.67 *** | -0.45 ** |
| Orange-winged parrot | --- | --- | --- | 1989-2006 (18) | 0.30 | 0.58 ** |

* = $P \leq 0.10$, ** = $P \leq 0.05$, *** = $P \leq 0.01$

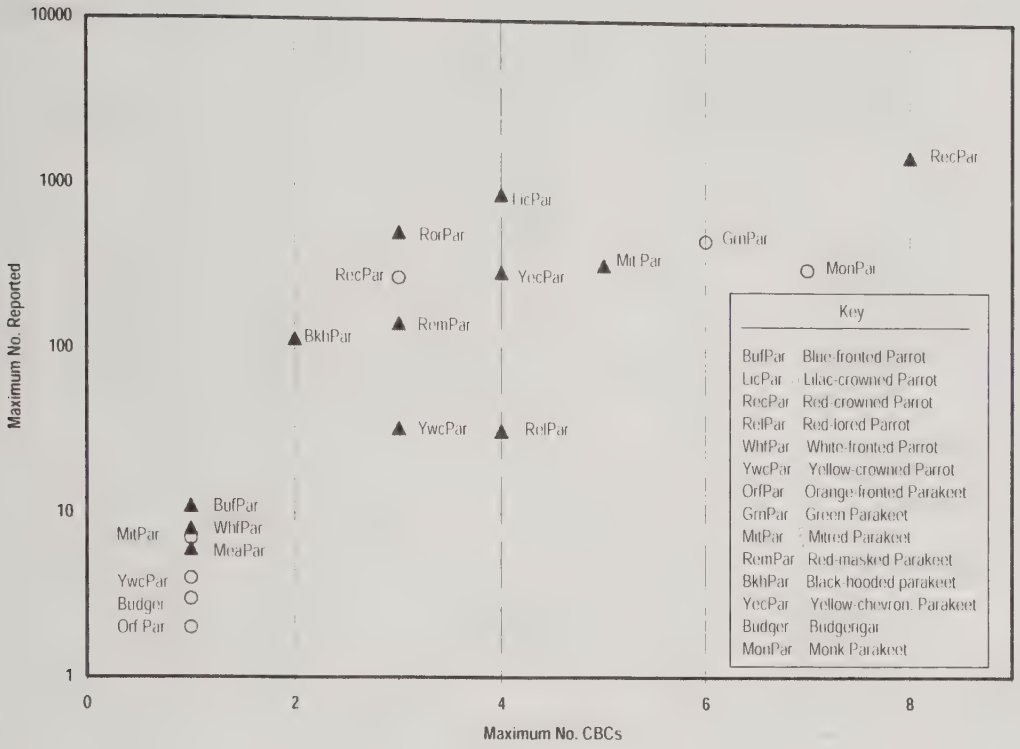



**Figure 4.** Distribution and abundance of 12 exotic parrots from CBC circles within California (solid triangles) and 7 species within Texas (open circles), 2000-06.

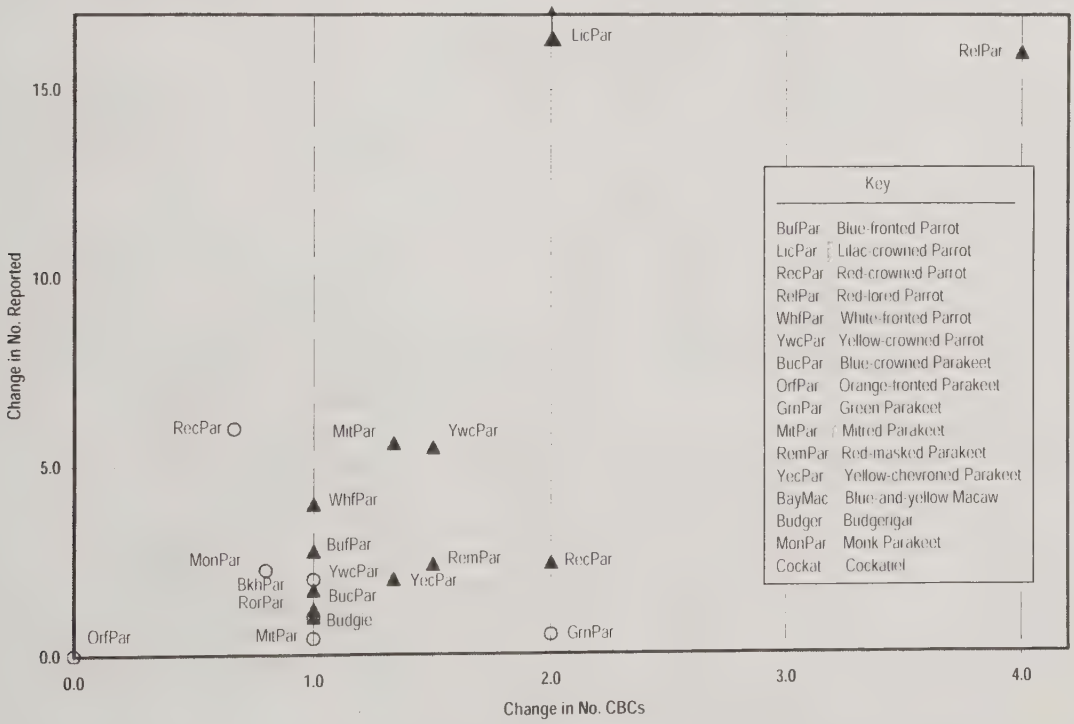



**Figure 5.** Changes in abundance and distribution of 16 exotic parrots from CBC circles within California (solid triangles) and 6 species in Texas (open circles), 2000-06.

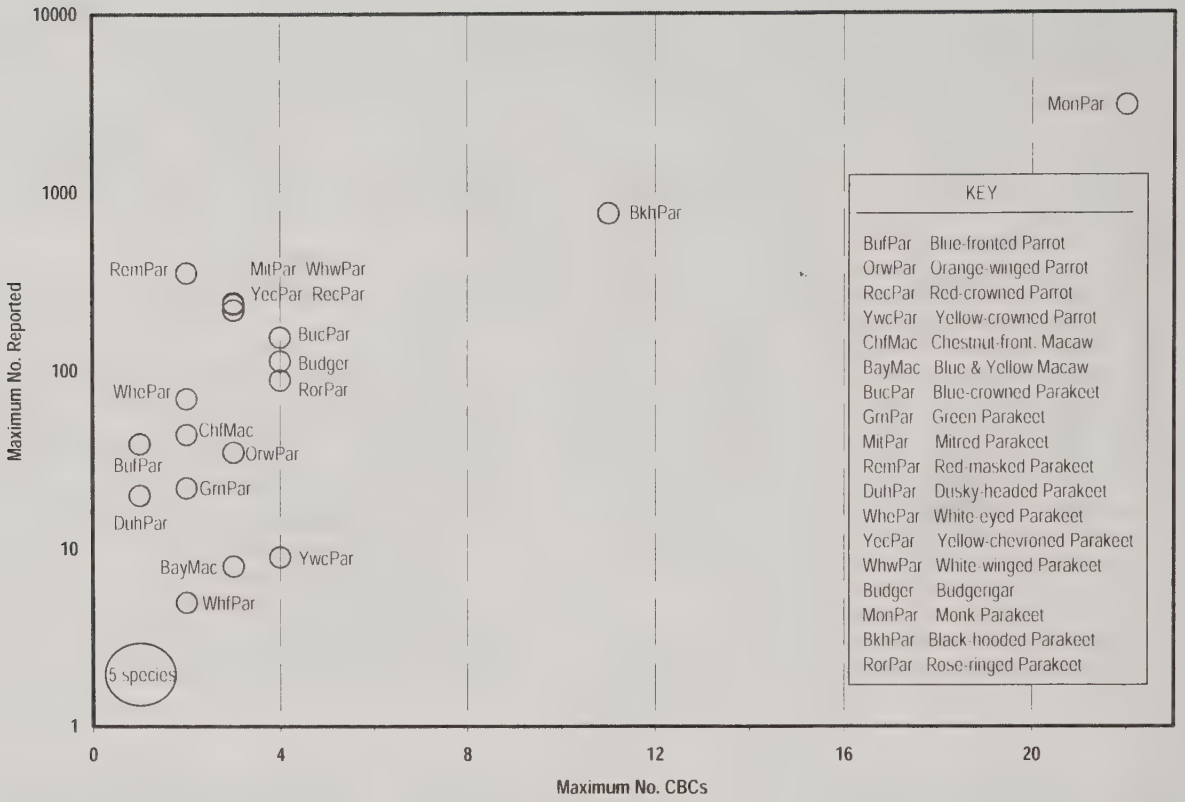



**Figure 6.** Distribution and abundance of 19 exotic parrots reported from CBC circles within Florida, 2000-06. Five additional species were reported from 1 count circle with <5 individuals: orange-fronted parakeet, red-lored parrot, lilac-crowned parrot, crimson-fronted parakeet, and cockatiel.

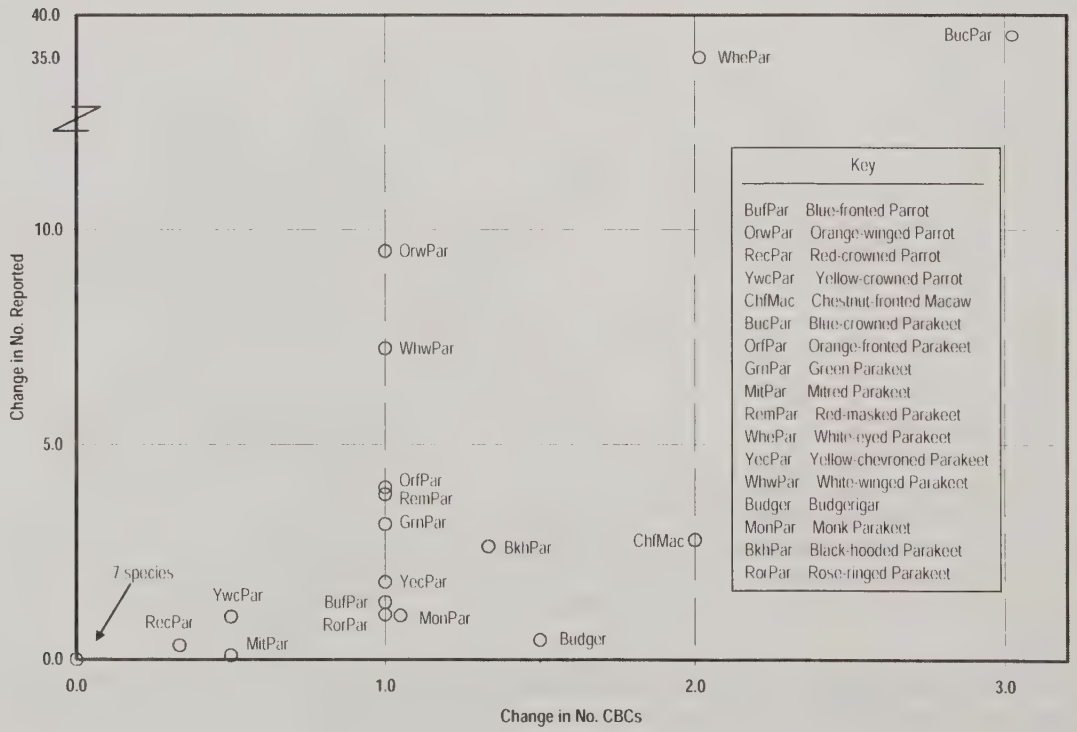



**Figure 7.** Changes in distribution and abundance of 18 exotic parrots from CBC circles within Florida, 2000-06. Seven additional species showed no net changes during this period: blue and yellow macaw, crimson-fronted parakeet, dusky-headed parakeet, lilac-crowned parrot, red-lored parrot, white-fronted parrot, and cockatiel.

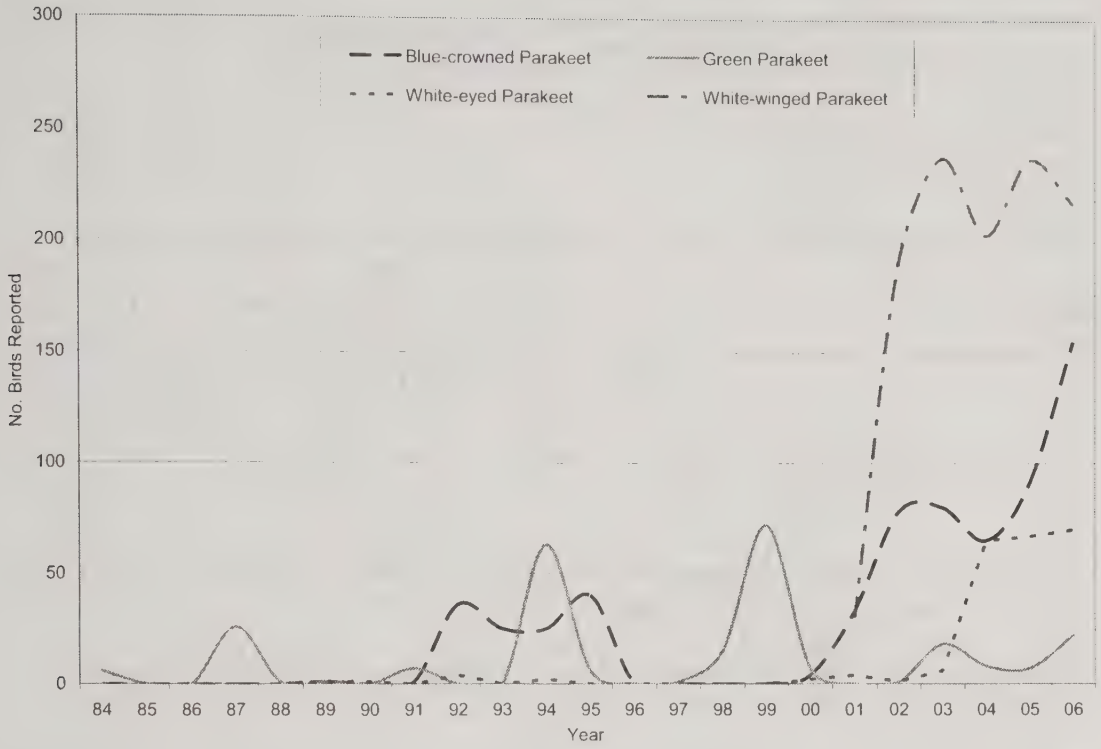



**Figure 8.** Trends in numbers of blue-crowned parakeets, green parakeets, white-eyed parakeets and white-winged parakeets from CBC circles within Florida, 1984-2006.

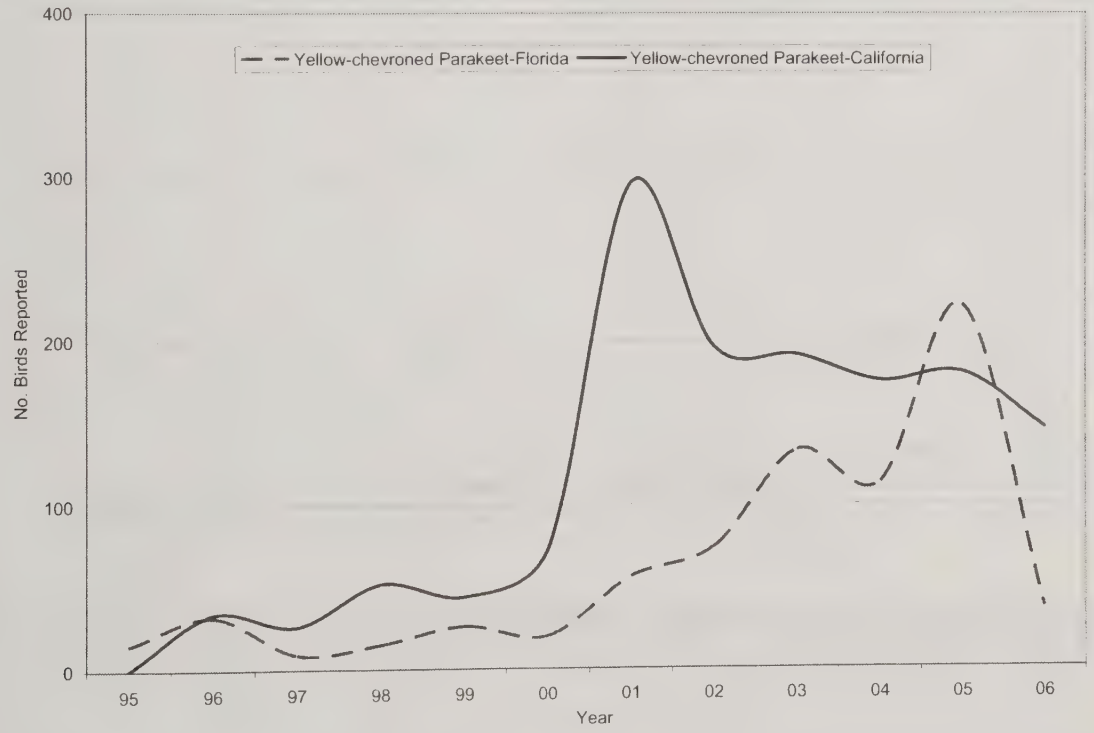



**Figure 9.** Trends in numbers of yellow-chevroned parakeets from CBC circles within California (solid line) and Florida (dashed line), 1995-2006.

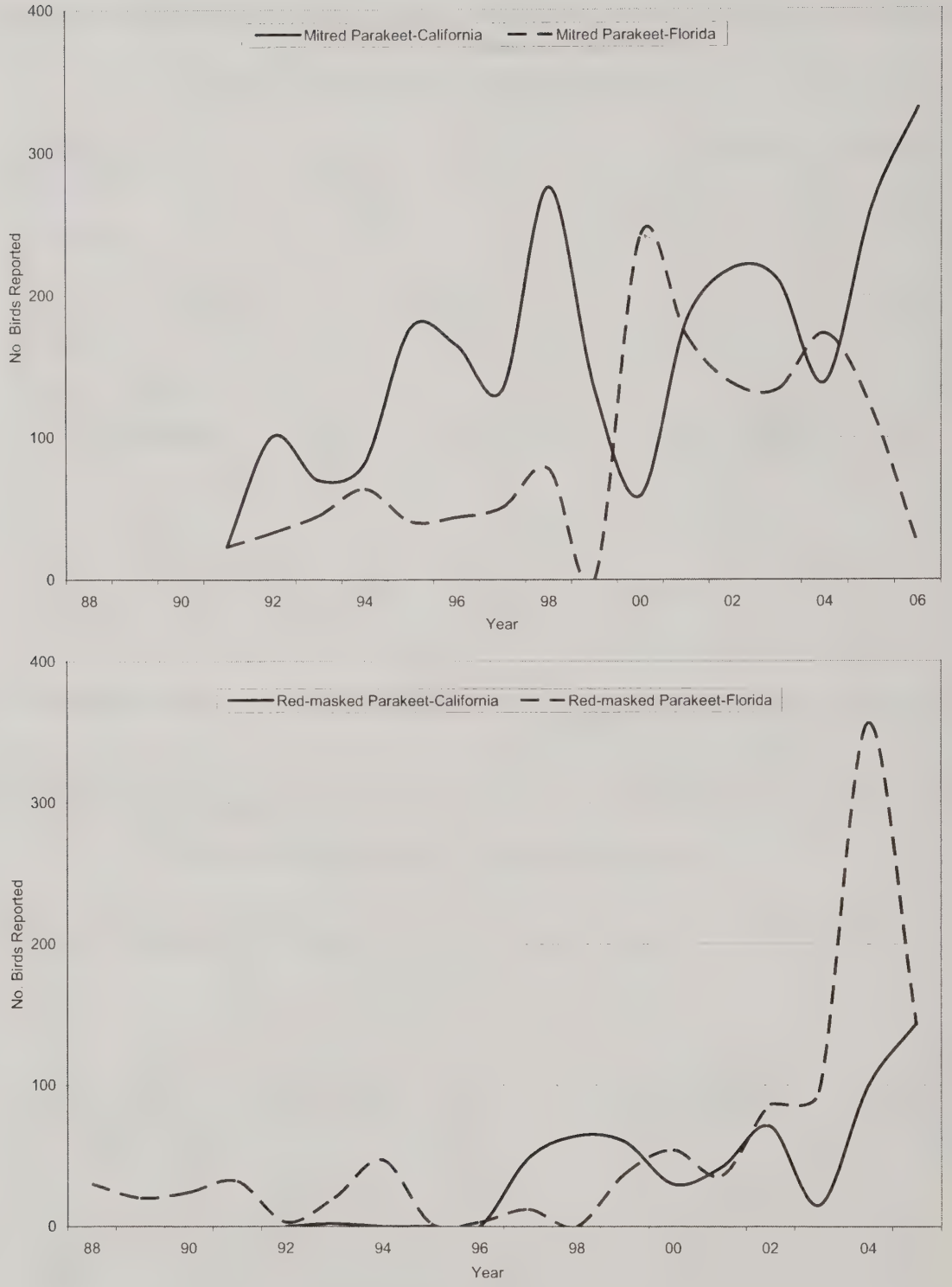



**Figure 10.** Trends in numbers of mitred parakeets and red-masked parakeets from CBC circles within California (solid line) and Florida (dashed line), 1988-2006.

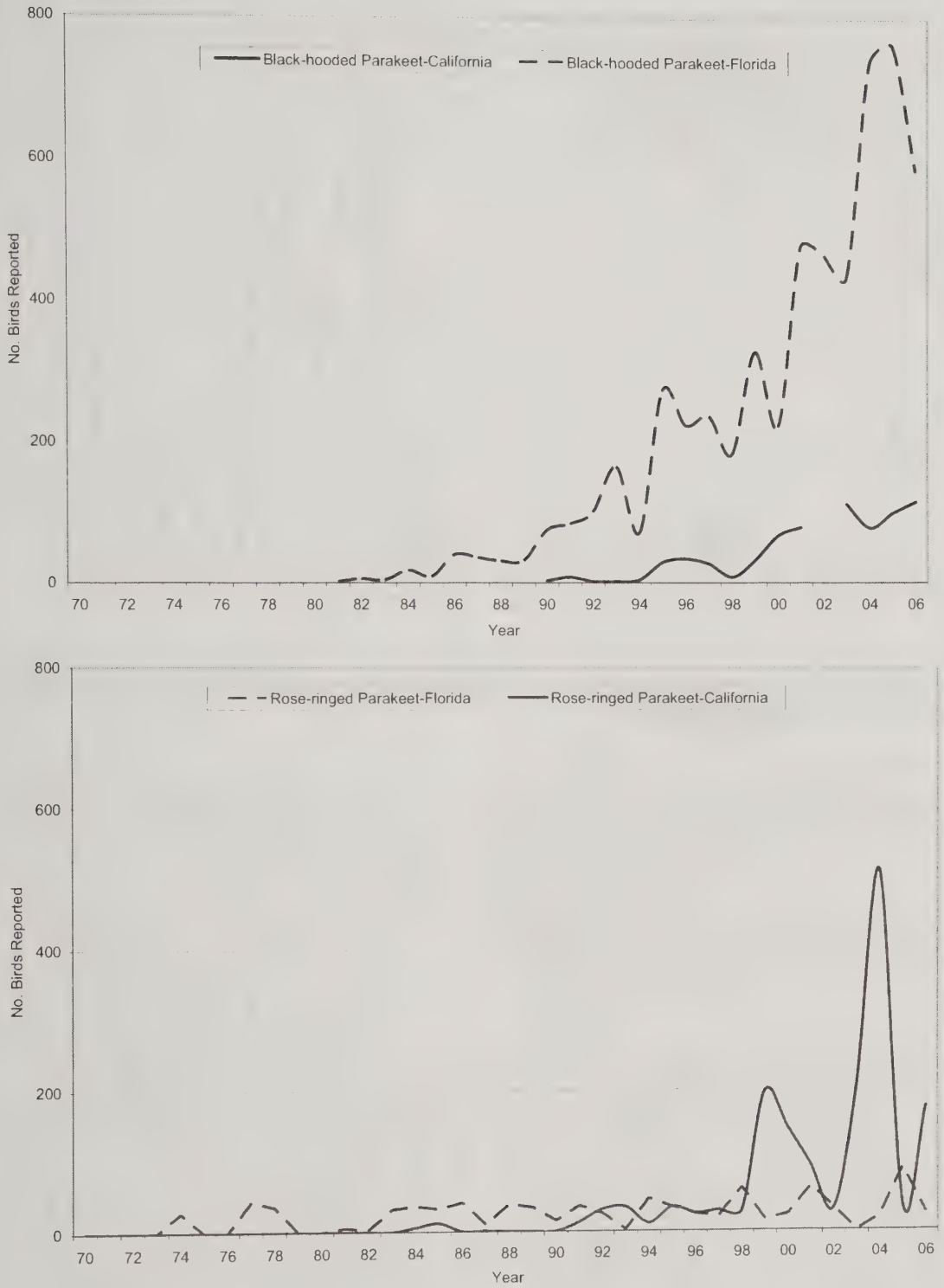



**Figure 11.** Trends in numbers of black-hooded parakeets and rose-ringed parakeets from CBC circles within California (solid line) and Florida (dashed line), 1970-2006.

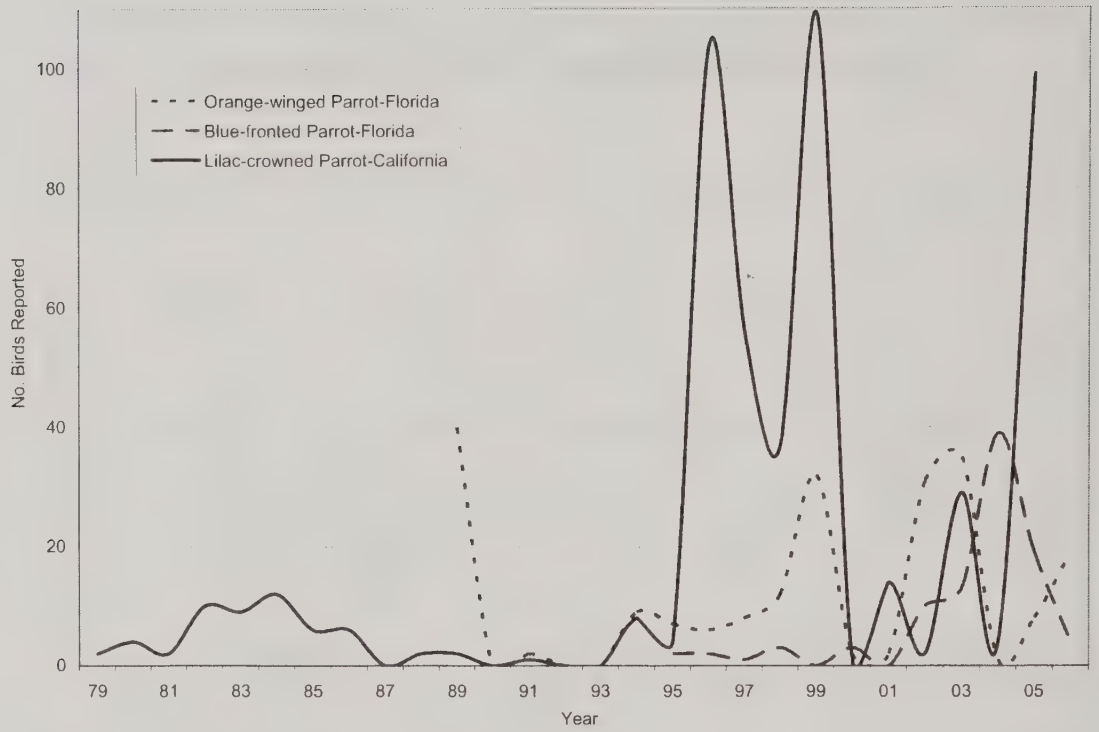



**Figure 12.** Trends in numbers of orange-winged parrots, blue-fronted parrots and lilac-crowned parrots from CBC circles within Florida and California, 1979-2005.

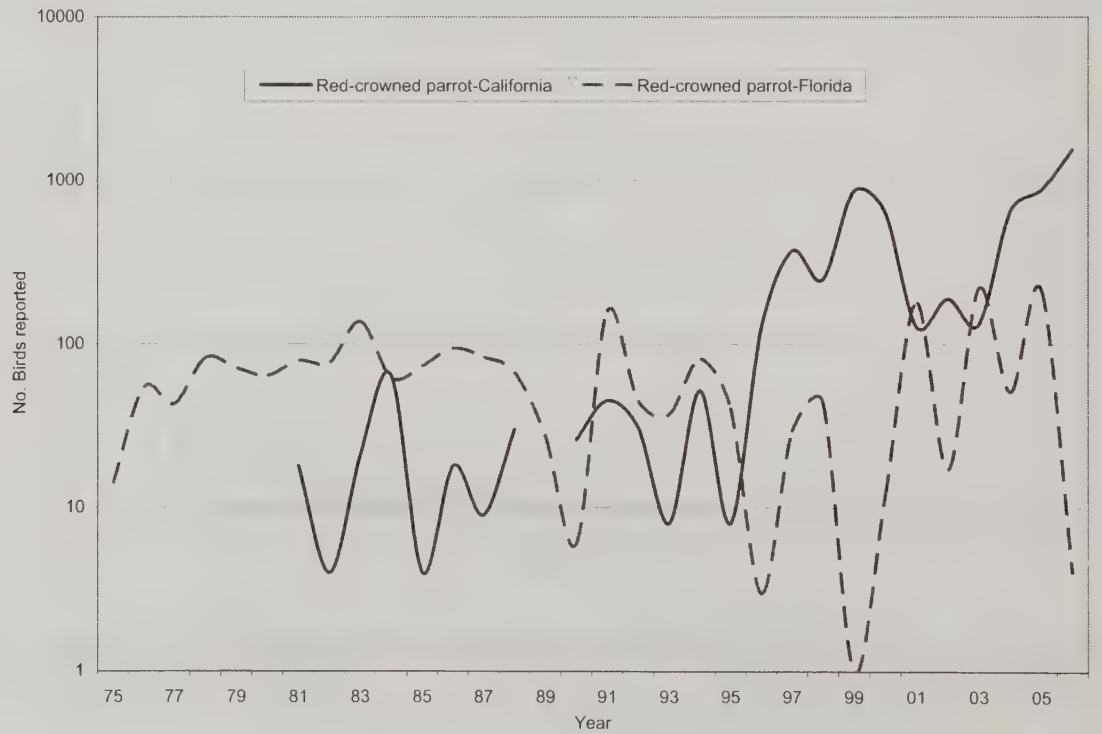



**Figure 13.** Trends in numbers of red-crowned parrots from CBC circles within California (solid line) and Florida (dashed line), 1988-2006.

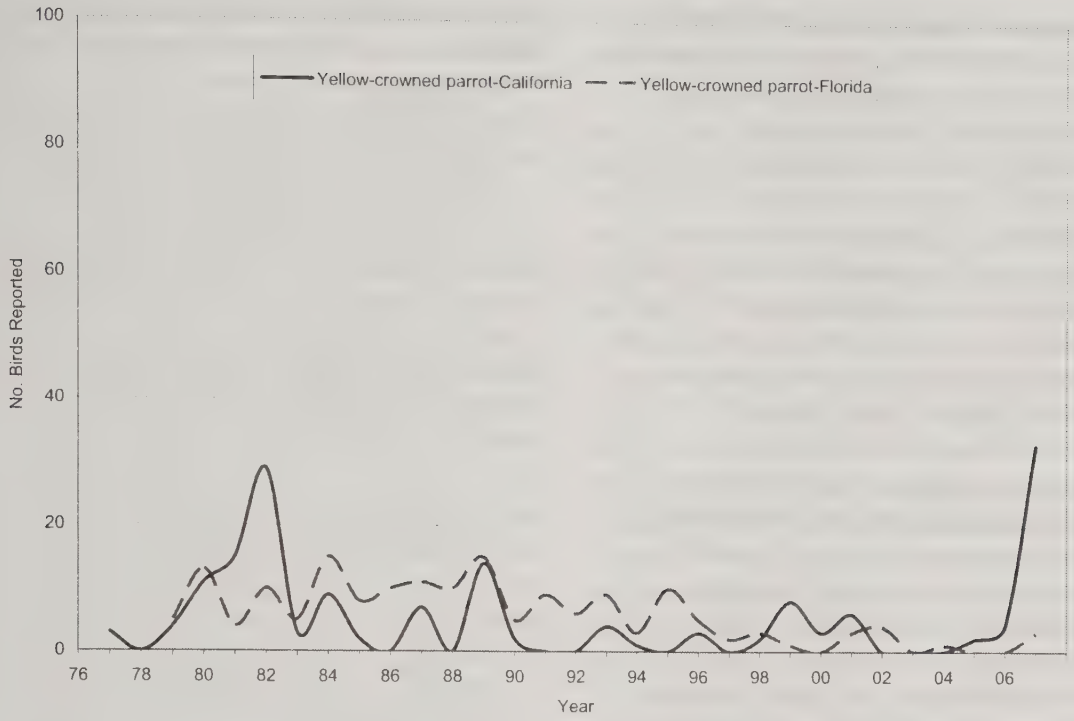



**Figure 14.** Trends in numbers of yellow-crowned parrots from CBC circles within California (solid line) and Florida (dashed line), 1976-2006.

in local *Brotogeris* parakeet populations. In his review of feral parrot populations, Butler (2005) described the status of several species on CBC circles within the US and identified five psittacines with significantly increasing populations: mitred, blue-crowned (*Aratinga acuticaudata*), and black-hooded parakeets, blue-fronted parrot (*Amazona aestiva*) , and chestnut-fronted macaw (*Ara severa*). Our results supported the idea that the three parakeets he identified have expanded in both numbers and distribution, and that the blue-fronted parrot population has increased (Table 5), but we could not confirm this for the chestnut-fronted macaw.

Population trends of monk parakeets have been analyzed with CBC data (Van Bael and Pruett-Jones 1996, Avery et al. 2002, Pruett-Jones et al. 2007) to show that the US and Florida populations of monk parakeets were increasing exponentially and doubling every six to seven years. Similar population expansions have been observed in Europe, where the species is also established (Sol et al. 1997). Plots of recent CBC data suggested that they may beginning to decline (Figure 1, 2) but long-term trends in abundance and distribution in Florida (Table 5) and across the US were positive ($R_s > 0.92$, $P < 0.0001$).

Monk parakeets are the best example of an invasive pest parrot in North America, but while distribution has increased over time (Figure 1) it remains limited. They have been reported from no more than 2.8% of all CBC circles across North America, but at the statewide scale they have been reported from up to 35% of the circles in Florida. To put these numbers in perspective, in recent decades, the invasive European starling has occurred on up to 96% of CBC circles across North America.

In contrast to the monk parakeet, the budgerigar population, which was deliberately established after repeated releases of up to 3,000 birds along the central Gulf Coast of Florida, did not become invasive (Figure 2). In the late-1970s, total numbers in the wild peaked at >20,000 birds. The population was concentrated in five counties along the Gulf Coast, but included birds widely scattered across 31 of Florida's 67 counties (Pranty 2001).

Budgerigars then declined dramatically in both abundance and distribution during the 1980s and 1990s, and recently numbered <200 birds in just two counties (Wenner and Hirth 1984, Pranty 2001, NAS 2002, Florida Fish and Wildlife Conservation Commission 2003). Pranty (2001) and Butler (2005) reviewed the potential causes for this population crash, but data are sparse and no firm conclusion has been presented. A breeding population on Tresco Island in the United Kingdom (UK) followed a similar track: in less than 10 years,

10 pairs increased to 100 birds then crashed and vanished after the resident who fed the birds moved away (Butler 2005). Rank correlation analyses for budgerigars reflected the dramatic decline in numbers from CBC circles within Florida, but the decline in distribution (Figure 2) was not significant (Table 5). An intermediate example (Figure 2c) is the recently established black-hooded parakeet population whose limited range is expanding in Florida (NAS 2002, Pranty and Lovell 2004). They are not currently considered invasive, but their numbers appear to be on the same trajectory as the monk parakeet (Figure 3).

Our calculations and graphs were designed to compare the relative abundance and distribution status of exotic populations of parrots in order to identify species that have increased rapidly in numbers and distribution. Our comparisons were relative and restricted to psittacines reported from CBC circles and the distribution of most species was relatively limited. In California, none occurred on more than 8 count circles. For Texas, the maximum number of CBCs was 7 seven circles (Figure 4). Similarly for Florida, only two species (monk and black-hooded parakeets) were reported from more than 4 CBCs (Figure 6).

The graphic summaries of recent (1999-2006) data from CBC circles within Florida, California, and Texas highlighted the following species as relatively widespread and abundant: monk parakeet, black-hooded parakeet, red-crowned parrot, green parakeet (*Aratinga holochlora*) , and mitred parakeet (Figures 4, 6). These 5 species were previously recognized as established by experts in the US (Table 2). The most rapidly expanding and increasing species in these years were the blue-crowned parakeet, white-eyed parakeet (*Aratinga leucophthalmus*), lilac-crowned parrot (*Amazona finschi*) and red-lored parrot (*Amazona autumnalis lilcana*; Figures 5, 7) which may be in the process of establishing naturalized populations. In addition to these, longer-term trends in numbers indicate that the yellow-chevroned and white-winged parakeet (*Brotogeris versicolurus*) are capable of rapid, if not sustained, population increases (Figures 8, 9).

As predicted by an exponential population growth curve, many exotic vertebrate populations have shown lag phases in population growth after introduction (Mack et al. 2000). After initial releases, introduced populations may first grow slowly as they build momentum and then suddenly irrupt. Following releases in Hawai'i, several currently naturalized birds were not found breeding or even sighted for many years (e.g., barn owls (*Tyto alba*), white-rumped shama (*Copsychus malabaricus*), and Guam swiftlets (*Aerodramus bartschi*; Foster 2005). Populations of the common myna and red-billed leiothrix both experienced a slow building period followed by rapid growth in Hawai'i (Foster 2005). CBC data reflected a similar pattern for several introduced parrot populations as well. Examples from Florida included monk parakeets (Figures 2, 3) and white-eyed parakeets (Figure 8) whose numbers built slowly for >10 years. In comparison, red-masked parakeet counts remained low for about 15 years (Figure 10). Black-hooded parakeets were recently accepted as an established part of Florida's avifauna following a lag of about 15 years (Figures 2, 3, 11). Counts of several *Amazona* spp. fluctuated greatly but have remained low for 10-20 years (Figures 12, 13). Similarly, Maui's mitred parakeet population, which was founded by a single pair, took 15 years to reach 200 (Runde and Pitt 2006). The lag periods for south Florida's yellow-chevroned and white-winged parakeets appeared short at about 5 years (Figures 8, 9), but both populations were founded by large-scale escapes of wild birds in the pet trade (Brightsmith 1999).

Lag periods for rose-ringed parakeets appear to be on the scale of decades. Numbers on California CBCs remained <200 for >20 years and still appeared low after 30 years in Florida (Figure 11). Established populations of rose-ringed parakeets in the UK appeared to grow slowly from 1969-1996, but have increased exponentially since then. Annual increases have recently been estimated at 15 to 30%, but range expansion at 300-400 m/yr has been relatively slow (Butler 2003). Similar growth rates were noted in the Netherlands, where the population has increased at a rate of 22.5% per year since 1994 (Keiji 2001). For long-lived parrots with low reproductive rates, this phenomenon of long lag periods preceding exponential population growth makes it especially difficult to determine if feral populations will become established or invasive and argues for timely control of incipient populations even though they may appear stagnant for 20-30 years.

## POTENTIAL IMPACTS OF EXOTIC PARROTS

Established populations of exotic parrots may pose problems to Hawai`i's natural systems as well as human health and safety, and agriculture. Since there are no native psittacines, emerging parrot populations may present novel threats to Hawai`i's ecology in ways that previous bird introductions have not. Although most psittacines are seed predators and are not generally recognized as seed dispersers, they do feed on many exotic invasive plants and there is concern that they may disperse viable seeds and over large areas. For example, mitred parakeets are known to feed on *Miconia calvescens* berries in their home range and captive feeding trials on Maui indicated that they can pass viable *Miconia* seeds (Gassmann-Duval 2002). and thus may currently be a factor in the long distance dispersal of invasive *Miconia* into native forests and other areas where much effort has been put into eradication. However, most parrots are seed predators feeding on immature seeds, thus the extent to which exotic parrots in general may disperse alien plants is unclear (Loope et al. 2001; J. Gilardi, personal communication).

As seed predators, parrots may inhibit reproduction of native tree species. Important foods for exotic parrots in Los Angeles included several palms (*Araceae* spp.; Garrett et al. 1997) and introduced sulfur-crested cockatoos (*Cacatua galerita*) reportedly fed on hearts of palm and devastated large stands of two endemic palms found only in Palau's rock islands (Engbring 1992, Loope et al. 2001, National Environmental Protection Council 2004). Thus, introduced psittacines may threaten Hawai`i's endangered loulu palms (*Pritchardia* spp.) and other native trees and shrubs with large seeds.

On Maui, direct competition between mitred parakeets and native birds has been observed in the displacement of frigate birds (*Fregata minor*) from roosting areas and in competition with shearwaters (*Puffinus* spp.) and petrels (*Pterodroma* spp.) for nesting sites in cliff-side burrows (MISC 2002) and may also be in conflict with Hawaiian black noddies (*Anous minutus*). A similar situation may exist with the burrowing parakeets on the Kona Coast. Roughly half of all psittacines include flower parts, nectar, or pollen in their diets and in Hawai`i, there is concern that these species could feed on native ohia trees (*Metrosideros polymorpha*) and other native flowering plants and might devastate native honeycreeper populations by competing for nectar resources (Loope et al. 2001).

A variety of parrot species are well-known agricultural pests in their native ranges typically targeting a wide variety of fruits, grains and oilseed crops (Bomford 1992). Habitats of about 30% of psittacines included orchards, plantations or croplands, but <4% included major Hawaiian crops such as coffee plantations or flooded taro fields. Cultivated tropical fruits, also important Hawaiian crops, were specifically included in summaries of diets for <10% of species; mango, citrus, banana and papaya were most frequent. Maize or various nut crops were included in diets of about 20% of species (T. Pratt and J. Foster, unpublished data). The agricultural impact of parrots in Hawai`i has not been assessed, but the potential for damage by existing species is evidenced by the fact that over 70% (17) of the 24 psittacines recently sighted in the state were identified as agricultural pest species, or as feeding on cultivated crops, orchards, etc. in their native ranges (Table 4). In a recent survey of farmers across Hawai`i, only 2% of respondents were concerned about damage from parrots to fruits and vegetables, such as corn, bananas, and papaya (NWRC, unpublished data). Our analyses suggest that the species most likely to emerge as agricultural pests in the US, if established, were the *Aratinga* parakeets and *Amazona* parrots of South America, cockatoos and corellas (*Cacatua* spp.) from Australia, lovebirds from Africa, and *Psittacula* parakeets from Africa and Asia.

The devastating impact of introduced parasites and disease on native Hawaiian forest birds has been well documented (Berger 1975; Van Riper et al. 1986, Van Riper and Scott 2001, Van Riper et al. 2002) and parrots can carry a variety of diseases that could impact native forest birds. As potential disease vectors, wide-ranging parrot populations threaten not only native bird species, but also poultry flocks, cage birds, aviculture, and human health. Diseases potentially transmissible to man or other birds include avian influenza, psittacosis (also known as ornithosis, chlamydiosis, and parrot fever), salmonellosis, pseudotuberculosis, tuberculosis, avian malaria, avian pox, erysipelas, pasteurellosis and Newcastle's disease. Of these, the first five are potentially of importance to human health. It has not yet been determined if any of the free-ranging parrots in Hawai`i carry these diseases. Since 2002, restrictions aimed at reducing imports of birds potentially infected with Newcastle disease and West Nile Virus into Hawai`i have been imposed. However, thousands

of birds have recently been imported into the State each year (Foster 2005). Currently, avian influenza if of great concern worldwide and there is widespread recognition of the risk of disease transmission associated with wildlife translocations and international trade in birds (Fevre et al. 2006). For highly pathogenic avian influenza (subtype H5N1), there is a potential for pandemic spread and cross-species transmission.

## CONCLUSION

Every invasive species was first introduced, and introduction effort is clearly the most important influence on the establishment of exotic birds in general, and of parrots in particular. Thus, minimizing the frequency of introductions, and the numbers of parrots released, will be the best way to prevent exotic parrots from becoming established in the future. This basic principle can be broadly applied; in their review of the status of a wide variety of exotic wildlife in North America and Europe, Jeschke and Strayer (2005) concluded, "...once a vertebrate is introduced, it has a high potential to become invasive. Thus, it is crucial to minimize the number of species introductions to effectively control invasive vertebrates". Because invasive species are rarely eradicated, and their damages will be borne for long periods, prevention is the best cure and will likely be the most cost-effective cure in the long run (Keller et al. 2007). As advocated by Loope et al. (2001), the list of birds permitted into Hawai'i should be reassessed to eliminate species with high risk of invasion. Past history of successful introduction, and trends in distribution and abundance of introduced populations are a reliable basis to identify parrots that pose a high risk of invasion. We used this along with three life history traits (extent of altitude range, diet breadth and degree of migration) to screen psittacines and identified 22 species which appeared most likely to establish naturalized breeding populations, if introduced. Currently, 17 of these are conditionally approved for import, meaning that permits may be granted for "individual possession, businesses, or institutions" (Hawai'i Department of Agriculture HRS Chapter 71). Policy makers might consider these as potential candidates for early detection surveillance, rapid response planning, or stricter import regulations.

## LITERATURE CITED


AMERICAN BIRDING ASSOCIATION (ABA). 2005. Checklist: birds of the continental United States and Canada. 6th edition. American Birding Association, Colorado Springs, Colorado, USA.

AMERICAN ORNITHOLOGISTS' UNION (AOU). 1998. Check-list of North American birds, 7th edition. American Ornithologists' Union. Washington D.C., USA. www.aou.org/checklist/ index.php3. Accessed January 10, 2007.

AVERY, M. L., E. C. GREINER, J. R. LINDSAY, J. R. NEWMAN, AND S. PRUETT-JONES. 2002. Monk parakeet management at electric utility facilities in South Florida. Pages 140-145 *in* R. M. Timm and R. H. Schmidt, editors. Proceedings of the 20th Vertebrate Pest Conference.

BANKS, R. C., C. CICERO, J. L. DUNN, A. W. KRATTER, P. C. RASMUSSEN, J. V. REMSEN, J. D. RISING, AND D. F. STOTZ. 2002. 43rd Supplement, American Ornithologists' Union checklist of North American birds. Auk 119:897-906.

BERGER, A. J. 1975. History of exotic birds in Hawaii, part II. `Elepaio 35:72-80.

BERGER, A. J. 1981. Hawaiian birdlife. 2nd edition. University of Hawaii, Honolulu, Hawaii, USA.

BISHOP, M. A. AND G. F. BENNETT. 1992. Host-parasite catalogue of the avian haematozoa Supplement 1. Memorial University of Newfoundland Occas. Pap. Biol. No. 15.

BLACKBURN T. M., AND R. P. DUNCAN. 2001a. Establishment patterns of exotic birds are constrained by non-random patterns in introduction. Journal of Biogeography 28:927-939

BLACKBURN T. M., AND R. P. DUNCAN. 2001b. Determinants of establishment success in introduced birds. Nature 414:195-197.

BOMFORD, M. 1992. Review of research on control of bird pests in Australia. Pages 93-96 *in* J. E. Borrecco and R. E. Marsh, editors. Proceedings of the 15th Vertebrate Pest Conference.

BRIGHTSMITH, D. 1999. White-winged parakeet (*Brotogeris versicolurus*) and yellow-chevroned parakeet (*Brotogeris chiriri*). No. 386-387 *in* A. Poole and F. Gill, editors. The birds of North America, Philadelphia, PA, USA.

BUTLER, C. J. 2003. Population biology of the introduced rose-ringed parakeet *Psittacula krameri* in the UK. Thesis, University of Oxford, UK.

BUTLER, C. J. 2005. Feral parrots in the continental United States and United Kingdom: past, present, and future. Journal of Avian Medicine and Surgery 19:142-149.

CALIFORNIA BIRD RECORDS COMMITTEE. 2006. California bird list. Western Field Ornithologists. www.wfo-cbrc.org/cbrc/ca_list.html.

CASSEY, P. T. 2002. Life history and ecology influences establishment success of introduced land birds. Biological Journal of the Linnean Society 76:465–480.

CASSEY, P. T., T. M. BLACKBURN, R. P. DUNCAN, AND K. J. GASTON. 2005. Causes of exotic bird establishment across oceanic islands. Proceedings of the Royal Society B: Biological Sciences 272:2059-2063.

CASSEY, P. T., M. BLACKBURN, K. E. JONES, AND J. L. LOCKWOOD. 2004a. Mistakes in the analysis of exotic species establishment: source pool designation and correlates of introduction success among parrots (Aves: Psittaciformes) of the world. Journal of Biogeography 31:277-284.

CASSEY, P. T., M. BLACKBURN, G. J. RUSSELL, K. E. JONES, AND J. L. LOCKWOOD. 2004b. Influences on the transport and establishment of exotic bird species: An analysis of the parrots (Psittaciformes) of the world. Global Change Biology 10:417-426.

CASSEY, P. T., M. BLACKBURN, D. SOL, R. P. DUNCAN, AND J. L. LOCKWOOD. 2004c. Global patterns of introduction effort and establishment success in birds. Proceedings of the Royal Society B: 271:S405-408.

DEL HOYO J., A. ELLIOTT, AND J. SARGATAL. 1997. Handbook of the birds of the world. Sandgrouse to cuckoos. Birdlife International, Cambridge, UK.

DIDHAMA, R. K., J. M. TYLIANAKIS, M. A. HUTCHISON, R. M. EWERSC, AND N. J. GEMMELL. 2005. Are invasive species the drivers of ecological change? Trends in Ecology and Evolution 20:470-474.

DOMÈNECH, J., J. CARRILLO AND J. C. SENAR. 2003. Population size of the monk parakeet *Myiopsitta monachus* in Catalonia. Revista Catalana d'Ornitologia 20:1-9.

ENGBRING, J. 1992. Surveys of forest birds of the Republic of Palau. U.S. Fish and Wildlife Service, Honolulu, Hawaii, USA.

EGUCHI, K., AND H. E. AMANO. 2004. Spread of exotic birds in Japan. Ornithological Science 3:3-11.

FEVRE, E. M., B. M. DE C. BRONSVOORT, K. A. HAMILTON, AND S. CLEAVELAND. 2006. Animal movements and the spread of infectious diseases. Trends in Microbiology 14:125-131.

FLORIDA FISH AND WILDLIFE CONSERVATION COMMISSION. 2003. Florida's breeding bird atlas: a collaborative study of Florida's birdlife. www.myfwc.com/bba.

FLORIDA ORNITHOLOGICAL SOCIETY RECORDS COMMITTEE. 2005. Official state list of the birds of Florida. www.fosbirds.org/RecordCommittee/StateListFebruary2005.htm. Accessed January, 8 and 10, 2007.

FORSHAW, J. M. 1977. Parrots of the world. Second edition. T. F. H. Publications, Neptune, New Jersey, USA.

FOSTER, J. F. 2005. Exotic bird invasion into forests of Hawaii: demography, competition, and seed dispersal. Thesis, University of Illinois, Urbana, Illinois, USA.

FOSTER, J. T. In Press. The history, naturalization, and impacts of introduced birds in the Hawaiian islands. Chapter 13 *in* T. Pratt, editor. Hawaii forest birds, University of Hawaii, Honolulu, Hawaii, USA.

GARRETT, K. L, K. T. MABB, C. T. COLLINS, AND L. M. KARES. 1997. Food items of naturalized parrots in southern California. West. Birds 28:196-201.

GASSMANN-DUVALL, R. 2002. Naturalized mitred conures on Maui and other parrots–pest evaluation and management proposal for non-native, invasive bird species. Unpublished draft report, Maui Invasive Species Committee, Makawao, Maui, Hawai'i, USA.

JESCHKE, J. M., AND D. L. STRAYER. 2005. Invasion success of vertebrates in Europe and North America. Proceedings of the National Academy of Sciences 102:7198-7202.

JUNIPER, T., AND M. PARR. 1998. Parrots. A guide to the parrots of the world. Yale University, New Haven, Connecticut, USA.

KEIJI, G. O. 2001. Ring-necked parakeets *Psittacula krameri* in Amsterdam, 1976-2000. Limosa 74:29-32.

KELLER, R. P., D. M. LODGE, AND D. C. FINNOFF. 2007. Risk assessment for invasive species produces net bioeconomic benefits. Proceedings of the National Academy of Sciences 104:203-207

KOLAR C. J., AND D. M. LODGE. 2001. Progress in invasion biology: predicting invaders. Trends in Ecology and Evolution 16:199-204.

LAMONT, D. 2002. Rainbow lorikeet. Farmnote No. 8/2002. Department of Agriculture West. Australia. Delivery Centre, Western Australia.

LEVER, C. 1987. Naturalized birds of the world. John Wiley & Sons, New York, NY, USA.

LOCKWOOD, J. L. 1999. Using taxonomy to predict success among introduced avifauna: relative importance of transport and establishment. Conservation Biology 13:560-567.

LONG, J. L. 1981. Introduced birds of the world. The worldwide history, distribution and influence of birds introduced to new environments. David & Charles, Newton, Abbot, London.

LOOPE, L. L., F. G. HOWARTH, F. KRAUS, AND T. K. PRATT. 2001. Newly emergent and future threats of alien species to pacific birds and ecosystems. Studies in Avian Biology 22:291-304.

MACK, R. N., D. SIMBERLOFF, W. M. LONSDALE, H. EVANS, M. CLOUT, AND F. A. BAZZAZ. 2000. Biotic invasions: causes, epidemiology, global consequences, and control. Ecological Applications 10:689-710.

MAUI INVASIVE SPECIES COMMITTEE (MISC). 2002. Action plan. Makawao, Hawai'i, USA. www.hear.org/misc/pdfs. Accessed January, 2007.

MOULTON, M. I., K. E. MILLER, AND E. A. TILLMAN. 2001. Patterns of success among introduced birds in the Hawai'ian islands. Studies in Avian Biology 22:31-46.

National Audubon Society (NAS). 2002. The Christmas bird count. Historical results. www.audubon.org/bird/cbc. Accessed October, 2006.

National Environmental Protection Council. 2004. Palau national invasive species strategy. www.palau.biodiv-chm.org. Accessed February 12, 2007.

Pranty, W. 2001. The budgerigar in Florida: rise and fall of an exotic *Psittacid*. American Birds 55:389-397.

Pranty, W. 2002. The use of Christmas bird count data to monitor populations of exotic birds. American Birds 56:24-28.

Pranty, W., and H. W. Lovell. 2004. Population increase and range expansion of black-hooded parakeets in Florida. Florida Field Naturalist 32:129-137.

Pranty, W., and K. L. Garrett. 2003. The parrot fauna of the ABA area: a current look. Birding 35:248-261.

Pratt, H. D., L. Bruner, and D. G. Berrett. 1987. A field guide to the birds of Hawaii and the tropical Pacific. Princeton University, Princeton, New Jersey, USA.

Pruett-Jones, S., J. R. Newman, C. M. Newman, M. L. Avery and J. R. Lindsay 2007. Population viability analysis of monk parakeets in the United States and examination of alternative management strategies. Human-Wildlife Conflicts 1:35–44.

Pyle, R. L. 2002. Checklist of the birds of Hawaii. Elepaio 62:137-148.

Ricciardi A., W. W. M. Steiner, R. N. Mack, and D. Simberloff. 2000. Computers in biology: toward a global information system for invasive species. BioScience 50:239-244.

Roe, D., T. Mulliken, S. Milledge, J. Mremi, S. Mosha, and M. Grieg-Gran. 2002. Assessing the impacts of wildlife trade regulations. Pages 32-54 *in* Making a killing or making a living? Wildlife trade, trade controls and rural livelihoods. Biodiversity and Livelihoods Issues No. 6. Chapter 4. International Institute of Environmental Development, and TRAFFIC. www.traffic.org. Accessed January 11, 2007.

Runde, D. E. and W. C. Pitt. 2006. Developing capture and removal methods for mitred conures (*Aratinga mitrata*) on Maui: report and recommendations. Unpublished report, US Fish and Wildlife Service, Honolulu, Hawaii, USA.

Simberloff, D., Schmitz, D. C., Brown, T.C. 1997. Strangers in paradise: impact and management of nonindigenous species in Florida. Island Press, Washington D.C., USA.

Simberloff, D., and L. Gibbons. 2003. Now you see them, now you don't! - population crashes of established introduced species. Biological Invasions 6:161-172.

Sol, D., and L. Lefebvre. 2000. Behavioural flexibility predicts invasion success in birds introduced to New Zealand. Oikos 90:599-605.

Sol, D., D. M. Santos, E. Feria, and J. Clavell. 1997. Habitat selection by the monk parakeet *Myiopsitta monachus* during the colonization of a new area. Condor 99:39-46.

Spreyer, M. R., and E. H. Bucher. 1998. Monk parakeet (*Myiopsitta monachus*). No. 322 *in* A. Poole and F. Gill, editors. The Birds of North America, Philadelphia, Pennsylvania, USA.

Texas Bird Records Committee. 2006. Accepted Texas species. Texas Ornithological Society. www.texasbirds.org/tbrc/statelis.htm.

Van Bael, S., and S. Pruett-Jones. 1996. Exponential population growth of monk parakeets in the United States. Wilson Bull 108:584-588.

Van Riper, C., III, and J. M. Scott. 2001. Limiting factors affecting Hawaiian native birds. Studies in Avian Biology 22:221-233.

Van Riper, C., III, S. G. Van Riper and W. R. Hansen. 2002. Epizootiology and effect of avian pox on Hawaiian forest birds. Auk 119:929-942.

Van Riper, C., III, S. G. Van Riper, M. L. Goff, and M. Laird. 1986. The epizootiology and ecological significance of malaria in Hawaiian land birds. Ecological Monographs 56:327-344.

Wenner, A. S., and D. H. Hirth. 1984. Status of the feral budgerigar in Florida. Journal of Ornithology 55:214-219.

Williamson, M. 1996. Biological invasions. Chapman & Hall, London, UK. Accessed January 22, 2007.

World Wildlife Fund. 2007. Parrot trade. Online article. www.worldwildlife.org/trade/faqs_parrot.cfm.

# MANAGING STARLINGS IN AUSTRALIA – CAN DRC-1339 BE THE ANSWER?

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***Abstract:*** European starlings (*Sturnus vulgaris*) are one of the most abundant and widespread bird species on earth. Introduced to Australia in the late 1850s, their highly adaptive ecology has enabled them to become a major pest. Concerns include public health and safety, economic impacts, detrimental environmental effects, and bird strikes at airports. Unfortunately current Australian control methods are ineffective and more efficacious, humane methods need to be developed. A feasibility study was undertaken of the likely risks/benefits of avicide DRC 1339 (3-chloro-4-methylbenzenamine hydrochloride) in Australia. This included a risk-analysis, a multi-industry survey and non-toxic bait trials to assess target-specificity. Results indicated that European starlings were one of three key bird species affecting intensive agriculture; there was strong support for better bird control techniques; and the risk to 82% of the native bird species recorded at the test sites was assessed as low or very low. Potential bait carrier trials demonstrated behavioral differences between United States and Australian starlings and hence the main application will initially be restricted to sites such as intensive livestock production facilities where starlings are already accustomed to feeding. Therefore, the project now aims to demonstrate DRC 1339 efficacy under Australian conditions and achieve registration.

***Key Words:*** 3-chloro-4-methylbenzenamine hydrochloride, Australia, avicide, bait trials, DRC-1339, invasive species, risk analysis, starlings, *Sturnus vulgaris*.

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## INTRODUCTION

European starlings have been classified as one of the top 100 worst invasive species worldwide (Lowe et al. 2000). They are a major pest in Australia (Figure 1) in that they cause significant economic damage to a variety of primary production industries (e.g., viticulture, horticulture, and livestock). They also have a significant negative impact on biodiversity and a marked social impact in urban environments. Starlings are further known to impact on human health and safety. The problems they cause include:

- Disease transmission (to animals and humans);
- Feed and water contamination;
- Droppings (nuisance, spreading weeds, contaminating wool);
- Impacting on natural biodiversity (overtaking nesting hollows and spreading weeds); and
- Economic losses (feed and crop consumption, clean-up costs from droppings, weed removal).

Due to their gregarious nature, extremely large flocks can occur throughout much of the year. Such aggregations of birds often foul public facilities when roosting overnight. Pairs and their young sully roofs with mites and dirty nesting material when flocks break up during the breeding season, also creating fire hazards. Contamination of areas with accumulated starling droppings poses a health risk, such as fouling rainwater used for drinking. Weber (1979) found over 25 important diseases can be carried by starlings, including bacterial, viral, mycotic, protozoal and numerous parasites, transferable to humans and livestock. The spores of *Histoplasmosis* are airborne, so starlings have the potential to affect much greater areas than just the primary site fouled. They have been known to be involved in bird strikes to aircraft. Starlings also are of detriment to native fauna and flora in that they disperse weeds and compete with native fauna for food and nesting resources.

Starlings often eat and foul large quantities of feed intended for livestock. For example, a flock of 200 starlings may eat around 80 kg of grain per week and contaminate even more with their




**Figure 1.** Pilot study sites in relation to starling distribution in Australia.

droppings (New York State Department of Environmental Conservation 2004). In addition, their droppings may foul water intended for livestock which then has to be cleaned daily at the expense of farmer's time and increased water wastage; an issue with stricter water restrictions due to severe drought conditions in Australia. They also foul livestock bedding which may spread Salmonella and other diseases to animals, and contaminate sheep fleeces with their droppings while ox-pecking. DRC-1339 has been the single most effective means of reducing feed and animal losses by starlings in the United States livestock feeding operations (Besser et al. 1967). DRC-1339, however, has never been used in Australia.

## FEASIBILITY STUDY RESULTS

A postal survey of intensive livestock producers (piggeries and feedlot operators) and grain handling facilities identified starlings as one of the three main pest bird species that impact on their industries, the other two being introduced sparrows and native cockatoos (Lapidge et al. 2006). Various methods of control are currently used to help alleviate the impacts of pest birds in Australia, including scaring devices, netting and shooting. Many primary producers also opt to not undertake control despite suffering pest bird damage. Netting is the only technique that is reportedly effective, but implementation may be cost prohibitive.

The survey also indicated strong support for the development of more humane and cost-effective pest bird control techniques because current methods rely largely on shooting and use of unregistered chemicals (particularly inorganic insecticides) for poisoning birds. There was a widespread intention amongst feedlot and piggery operators to use poison bait if it was found to be cost effective, target-specific and humane. Acceptance of such a product was considerably less when horticulturists were asked the same question. A small, but significant, market potential therefore exists for the use of DRC-1339 in Australia, particularly in the intensive livestock industries. Interest from the Western Australian Government has also been registered.

Risk to a large majority (82%) of the Australian native bird species recorded at the test sites was assessed as low or very low and the toxin was assessed as highly humane (Dawes 2005, Dawes 2006). Lapidge et al. (2006) consequently trialed various bait substrates in three different climatic regions across Australia where starlings cause an impact. Figure 1 shows locations of: semi-arid pastoral Ceduna, SA (mean annual rainfall 290 mm), the Mediterranean climate wine-grape growing region of McLaren Vale, SA (mean annual rainfall 660 mm) and a temperate water waste facility in Orange, NSW (mean annual rainfall 940 mm). Trials of potential bait substrates confirmed the apparent difference between starling dietary behavior in Australia compared with the United States and, hence, the main application will initially be restricted to intensive livestock production facilities where feeding on non-live food has been well established.

Feeding trials in broadacre settings suggested that: (1) starlings are highly neophobic with respect to new food materials and foodstuffs presented in unaccustomed manners, and (2) other birds such as gulls and mynas may out-compete starlings for artificially-presented feed. The most universally accepted and starling-specific bait was raisins. Bread and dripping was another popular bait medium, but most (in New South Wales) to all (in South Austraila) of this material was taken by non-target species.

## DRC 1339 AND OFF TARGET SPECIES

The open use of DRC 1339 in Australia would put non-target species at risk, especially considering that no species-specific potential carrier bait substrate was found. In particular, high susceptibility to the active compound is indicated for members of the Corvidae (crows, ravens), Columbidae (doves, pigeons), Laridae (gulls), Phasianidae (quail) and Anatidae (ducks) bird families (Eisemann et al. 2002). As further

discussed below, contained use of DRC 1339 could minimize risk to potentially susceptible non-target species, while assisting significant 'high-intensity' primary production enterprises to reduce problems currently associated with pest birds.

Since the chemical is metabolized and its non-toxic metabolites excreted by starlings in about two hours, it is considered non-hazardous to predators or scavengers. Starlings that ingest the toxicant die one to three days after treatment, and dead birds are usually found at their roosting, loafing or feeding areas (DeCino et al. 1966). DRC 1339 also degrades rapidly when exposed to moisture, sunlight, heat or UV radiation (e.g., Tawara et al. 1996). It is environmentally safe in that it binds tightly to soils, has low mobility, degrades rapidly and will not migrate. The useful life of exposed baits can vary between a couple of hours when under high humidity and sunlight to more than a week under dark, dry conditions. Therefore, there is little potential for secondary hazards to non-target animals with DRC 1339.

Studies show that a Cooper's hawk (*Accipiter cooperii*), a marsh hawk (*Circus cyaneus*), and a sparrow hawk (*Falco sparverius*) that subsisted on field-killed DRC 1339 starlings for 3- and 4-month periods showed no ill effects (DeGrazio 1968). One report of secondary toxicity occurred when crows ate the gut contents of pigeons killed with Starlicide (see ACVM 2002 and Cummings et al. 2002). Cunningham et al. (1979) estimated that a sensitive species (i.e., cat, owl, and American magpie) could be at risk only if its diet consisted wholly of DRC 1339-poisoned starlings for more than 30 continuous days; however, risk is minimal because use and exposure to bird carcasses occurs for just under two weeks.

Techniques to reduce non-target bird hazards include limiting bait exposure on bait sites, diluting treated bait with untreated bait, positioning bait lanes away from field edges, and observing pre-baited fields to avoid those with non-target species present. Where starlings exist in high numbers, such as around piggeries, they often exclude all other species, hence preventing non-target birds from entering the area. The commercial set up of study sites such as piggeries, feedlots and grain bunkers, also tend to exclude native bird species. Another level of safety sometimes used is presenting baits in feeding trays with a thin wire mesh. This allows the slender beaks of starlings to reach the food whilst excluding parrots and other granivorous birds with short or rounded beaks. Additional precautions to be taken in future studies include searching for and collection of carcasses during baiting and the 5-day post-treatment period. Searches will be aided by locating flight paths and communal roost sites during the 10-day pre-treatment period.

Both starling neophobia/competition and non-target bird safety could potentially be resolved by pre-feeding starlings with accustomed feedstuffs at suitable baiting locations adjacent to regular feeding sites prior to deployment of the poisoned material. This approach is likely to be particularly applicable to sites of predictable high concentration feeding, such as piggeries and feedlots. Effective management of starlings at these sites would: (1) reduce a major pest bird impact on primary production in Australia, and (2) assist in reduction of recruitment and dispersal of replacement members for existing populations.

## PROJECT FUTURE

This project aims to demonstrate the feasibility and efficacy of using DRC 1339 for managing starlings under Australian conditions. The trials are planned to take place at intensive livestock production and grain-handling facilities where large numbers of starlings often congregate and cause significant economic losses. The chemical is widely used for this purpose in the US where, despite starlings being ground foraging insectivores, a range of non-live bait substrates can be used successfully to carry the chemical to the birds (e.g., DeGrazio 1968, Cummings et al. 2002). However, our earlier pilot project with non-toxic baits demonstrated that starlings in Australia will only take non-live baits at sites where they were already accustomed to feeding on similar foods, such as at intensive livestock and grain-handling facilities.

Trials will occur across a range of agricultural production sites (differing climates, livestock foods or grains, and different potential non-target bird issues), because it is necessary to submit data for product registration that demonstrates efficacy under a range of Australian conditions. It will also result in more efficacious and more humane pest bird management at these commercial facilities. The project is currently supported by the National Feral Animal Control Program (Natural Heritage Trust), Invasive Animals Cooperative Research Centre and Australian Pork Limited.

## RESEARCH BENEFITS

This project will improve the health and welfare for livestock and wildlife, the environment in general, and the community, through improved

starling management in Australia. DRC 1339 is used extensively in the US for the same purpose, but has not been tested in Australia. The project will determine whether or not the chemical is useful under Australian conditions. It will achieve this through critically assessing the efficacy and cost-per-starling-removed value as well as flow-on benefit for control techniques, including no control. This will provide new information, and will lead to, with appropriate extension activities and acceptance, improved starling management throughout Australia.

## LITERATURE CITED


ACVM. 2002. Controlled pesticides DRC-1339 for bird control. Prepared by the Agricultural Compounds and Veterinary Medicines Group for ERMA New Zealand 95.13 ACVM 07/02.

Besser, J. F., W. C. Royall, Jr., and J. W. DeGrazio. 1967. Baiting starlings with DRC-1339 at a cattle feedlot. Journal of Wildlife Management 31:48-51.

Cunningham, C .J., E. W. Schafer, and K. McConnell. 1979. DRC-1339 and DRC-2698 residues in starlings: preliminary evaluation of their effects on secondary hazard potential. Wildlife Damage Management, Internet Center for Bird Control Seminars Proceedings, University of Nebraska, Lincoln, Nebraska, USA.

Cummings, J. L., J. F. Glahn, E. A. Wilson, and J. E. Davis Jr. 2002. Potential hazards of DRC-1339 treated rice to non-target birds when used at roost staging areas in Louisiana to reduce local populations of depredating blackbirds. International Biodeterioration & Biodegradation 49:185–188.

Dawes, J. 2005. Risk assessment for the use of 3-chloro-p-toluidine hydrochloride to control starlings in Australia. Unpublished Pestat Ltd report. January 2005.

Dawes, J. 2006. Is the use of DRC-1339 (Starlicide®) humane? Unpublished Pestat Ltd report. March 2006.

DeCino, T. J., D. J. Cunningham, and E.W. Schafer Jr. 1966. Toxicity of DRC-1339 to starlings. Journal of Wildlife Management 30:249-253.

DeGrazio, J. W. 1968. DRC-1339 in feedlots. Wildlife Damage Management, Internet Center for Bird Control Seminars Proceedings, University of Nebraska, Lincoln, Nebraska, USA.

Eisemann, J. D., P. A. Pipas, and J. L. Cummings. 2002. Acute and chronic toxicity of compound DRC-1339 (3-chloro-4-methylalinline hydrochloride) to birds. Management of North American blackbirds. *In* G. M Linz, editor. Proceedings of a special symposium of The Wildlife Society 9th Annual Conference. Bismarck, North Dakota, USA.

Lapidge, S., D. Dall, J. Dawes, J. Tracey, R. Sinclair, and T. Bentz. 2006. A feasibility study of DRC-1339/Starlicide as an avicide in Australia. Unpublished National Feral Animal Control Program Report, Pestat Ltd, Canberra, Australia.

Lowe S., M. Browne, S. Boudjelas, and M. De Poorter. 2000. 100 of the world's worst invasive alien species: a selection from the global invasive species database. The Invasive Species Specialist Group (ISSG) a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN). First published as special lift-out in Aliens 12, December 2000. Updated and reprinted version November 2004.

New York State Department of Environmental Conservation. 2004. 3-chloro-4-methylbenzenamine hydrochloride (Starlicide) NYS DEC Letter – Product registration 1/04. Albany, New York, USA.

Tawara, J. N., J. J. Johnston, and M. J. Goodall. 1996. Degradation of 3-chloro-p-toluidine hydrochloride in watermelon bait. Identification and chemical characterization of novel *N*-glucoside and oxopropanimine. Journal of Agriculture and Food Chemistry 44:3983-3988.

Weber W.J. 1979. Health hazards from pigeons, starlings and English sparrows. Thomson Publications, Fresno, California, USA.

# FLORIDA'S NON-NATIVE AVIFAUNA

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***Abstract:*** Florida has a mild climate, diverse natural habitats, and a growing, mobile human population. Florida also hosts thousands of species of introduced plants, fish, and wildlife. A recent compilation lists 196 non-native bird species, comprising 15 orders, which have occurred in Florida. The list includes 72 species of parrots and parakeets (psittaciformes), 51 passerines, and 22 species of waterfowl (anseriformes). First sightings of newly introduced bird species in Florida increased rapidly through the 1980s, but numbers of new species detected appear to have subsided since then. Sources of introductions are often not reported, but of those that are known, most derive from intentional releases (e.g., for hunting) or escapes from private collections, parks, or zoos. Based on this list and other sources within the state, we identified 14 non-native bird species now considered to be established in Florida. We review those species in particular, and discuss impacts that they are having. We use Christmas Bird Count data to examine geographic and temporal patterns of non-native bird occurrence within the state. We identify additional non-native bird species that are emerging as potentially serious management problems in Florida, and we recommend actions for efficient, effective management of non-native bird issues.

***Key Words:*** Christmas Bird Count, Florida, invasive species, non-native birds, population trend.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

The house sparrow (*Passer domesticus*), a European native, has been in Florida since 1882, so non-native birds have been part of the Florida fauna for at least 125 years. Psittacines, escaped from captivity, have been flying around south Florida since the 1920s, possibly longer (Bailey 1928). The real expansion in numbers of non-native species occurred in the latter half of the 20th century, however, and was facilitated by a number of factors (Owre 1973). Natural habitats in south Florida were converted to accommodate increasing demands of the burgeoning human population for houses, businesses, and services. As the native landscape was transformed, hundreds of exotic plant species became established through horticulture or accidental release. Hundreds of thousands of exotic birds passed through the Miami airport. Many escaped there before getting to pet shops, zoos, and private exhibitors. Others were subsequently released on purpose or by accident. The mild year-round climate coupled with flora that was often identical to that found in their native range provided ideal conditions for free-flying non-native birds. Proliferation of backyard bird feeders enhanced the favorable environment for exotic birds.

The Wild Bird Conservation Act (1992) effectively prohibited importation of most wild-caught birds. There remains, however, thriving commerce in captive-reared exotic birds to supply the ever-present demands of the pet trade. Estimates of pet bird ownership in the United States (US) vary widely, from 10 million (Wise et al. 2002) to 40 million (www.avianwelfare.org/issues/overview.htm). Thus, even if all avian imports stopped immediately, there still would exist a huge, renewable pool of non-native birds to serve as a source for new free-flying exotics.

In this paper, we review the available information on the status of non-native birds in Florida. We examine patterns and trends of non-native birds as a component of the Florida avian population, and we discuss the nature of the problems posed by this segment of the Florida avifauna. Throughout, we apply the nomenclature of Executive Order 13112 "Invasive Species" published in February 1999. Thus, an "alien species" (or exotic or non-native) is a species not native to the ecosystem under consideration. An

"invasive species" is an alien species whose introduction is likely to cause harm, either economically, environmentally, or to human health. An "introduction" is the placement of a species into an ecosystem as a result of human activity. A "native species" is one that occurs in a particular ecosystem not as a result of an introduction.

## METHODS

### Information Sources

We obtained information on non-native birds from 3 main sources. On its website, the Florida Fish and Wildlife Conservation Commission (FWC) lists 196 non-native bird species that have been recorded in Florida (myfwc.com/nonnatives). According to the FWC website list, just 11 of these species are considered to be established, i.e. "...confirmed breeding and apparently self-sustaining for 10 or more consecutive years."

As of February 2007, the Florida Ornithological Society (FOS, www.fosbirds.org) recognized 498 species comprising the Florida avifauna, including 12 established exotics. The term established here means "... a stable or increasing population of that species has persisted continuously in one or more areas for at least 15 years", and there is "little or no evidence that ongoing releases play a significant role in population maintenance".

Finally, the compilation of breeding birds in Florida includes 165 native species, as well as 13 additional non-native species considered to be established (Woolfenden et al. 2006). Their criterion for establishment is "stable or increasing populations maintained by successful reproduction for at least 10 years up to the present". Furthermore, there are also 29 other non-native species on this list that are breeding in Florida, but which have not yet achieved the status of "established". This implies that a number of additional species might become established in the near future.

### Assessment of Status and Potential Threat

From the 3 major sources given above, we assembled a composite list of non-native Florida bird species, and we categorized the species according to their status and evaluated the potential threat they posed to natural and to man-made resources. This was our attempt to place in perspective the relative management importance of the established, non-native bird species in the state. We used the following categories:

- Ubiquitous, nuisance species – present throughout Florida, widespread impacts;
- Restricted range – common locally, not causing serious problems;
- Major problems – causing major ecological or monetary impact;
- Emerging threats – of concern, and still vulnerable to eradication;
- Formerly present – extirpated after once being established.

### Non-native Species Trends

We estimated the proportion of non-native birds in the Florida avifauna from Christmas Bird Count (CBC) data for selected sites and years (www.audubon.org/bird/cbc/index.html). To examine the pattern across the state, we calculated the percentage of non-native birds in the most recent count (#107, December 2006-January 2007) at CBC sites within Florida, representing 15 of the 17 most populous metro areas within the state. We then examined the trends from 1970-2005 at 2 Atlantic coast CBC sites, one in south Florida (Dade County) and one in north Florida (Jacksonville).

## RESULTS

### Non-native Species List

From the 3 sources we consulted, there was generally good agreement as to the roster of established species (Table 1; scientific names of bird species in the table are not repeated in the text). Probably the chief difference among the sources is that FOS considers the mallard and the white-winged dove to be native, and thus does not include these species on their list. Unlike the FWC and Woolfenden et al. (2006), however, the FOS considers the black-hooded parakeet to be established. The 3 sources of established exotic bird species are not independent as many of the same individual ornithologists contributed to the compilations. The 3 lists converge on a core set of 14 species that represent Florida's established non-native avifauna which we considered in more detail.

### Status and Potential Threat

*Ubiquitous, nuisance species.* It seems unlikely that species as well-entrenched as these could ever be eliminated from the avifauna. They are here to stay, and the best course of action is to monitor interactions so that when an unexpected impact is

**Table 1.** Non-native bird species considered established in Florida by various authorities: Florida Fish and Wildlife Conservation Commission (FWC), Woolfenden et al. (2006, WRC), and Florida Ornithological Society (FOS).

| Scientific Name | Common Name | FWC | WRC | FOS |
|---|---|---|---|---|
| *Cairina moschata* | Muscovy duck | x | x | x |
| *Anas platyrhynchos* | Mallard | | x | |
| *Columba livia* | Rock pigeon | x | x | x |
| *Streptopelia decaocto* | Eurasian collared-dove | x | x | x |
| *Zenaida asiatica* | White-winged dove | x | x | |
| *Melopsittacus undulatus* | Budgerigar | x | x | x |
| *Myiopsitta monachus* | Monk prakeet | x | x | x |
| *Nandayus nenday* | Black-hooded parakeet | | | x |
| *Brotogeris versicolurus* | White-winged parakeet | | x | x |
| *Pycnonotus jocosus* | Red-whiskered bulbul | x | x | x |
| *Sturnus vulgaris* | European starling | x | x | x |
| *Icterus pectoralis* | Spot-breasted oriole | x | x | x |
| *Carpodacus mexicanus* | House finch | x | x | x |
| *Passer domesticus* | House sparrow | x | x | x |

detected, remedial steps can be promptly taken. These species exploit human-altered environments and seldom venture into natural habitats. What impact are these birds having? The main impacts seem to be nuisance and aesthetics, and these are principally due to the big 3 of invasive birds – European starling, house sparrow and rock pigeon. All are ubiquitous in urban/suburban areas and each has become a fixture of the bird communities. The Muscovy is rapidly becoming the waterfowl equivalent of the rock pigeon in its fecundity, frequency of occurrence, and propensity to create unsavory, possibly unhealthy, conditions.

The house sparrow, one of the most common invasive bird species in the country, is exhibiting a long, strong decline, not only in Florida but throughout the US. According to Breeding Bird Survey (BBS) trend results, the Florida population has declined 70% to 75% in the past 40 years (Sauer et al. 2005). For the house sparrow nationwide, the BBS indicates a downward trend of approximately 60% in the same time frame. In Great Britain, a similar downward trend in the house sparrow population is attributed to changes in habitat (Robinson et al. 2005).

In the US, there is some evidence that competitive interactions with the house finch are contributing to the decline of the house sparrow (Cooper et al. 2007). The Florida situation, as evidenced by BBS data, clearly shows that both species are in the midst of trends, with the house sparrow heading down and the house finch going strongly upward (Figure 1). No cause-effect relationship has been established, however, so in Florida it is not clear how the burgeoning house finch population is affecting the house sparrow.

Many house finches carry a form of conjunctivitis that potentially could be spread to other wild bird species and to poultry (Luttrell et al. 2001, Farmer et al. 2005). The house finch frequents bird feeders and is well-suited to suburban and urban environments. As the house finch continues to expand throughout Florida, there will be increased concern for its role as carrier or reservoir of this disease.

In Florida, the Eurasian collared-dove and white-winged dove have the potential for negative impacts to native dove species. But, despite dietary overlap, there is as yet no evidence of a large-scale negative interaction with native mourning doves (*Zenaida macroura*) or any other species (Poling and Hayslette 2006). For example, the mourning dove count on the BBS continues to increase slowly and steadily even as numbers of the non-native doves increase at much faster rates and overtake the mourning dove in terms of number of birds counted (Figure 2). The Eurasian collared-dove particularly

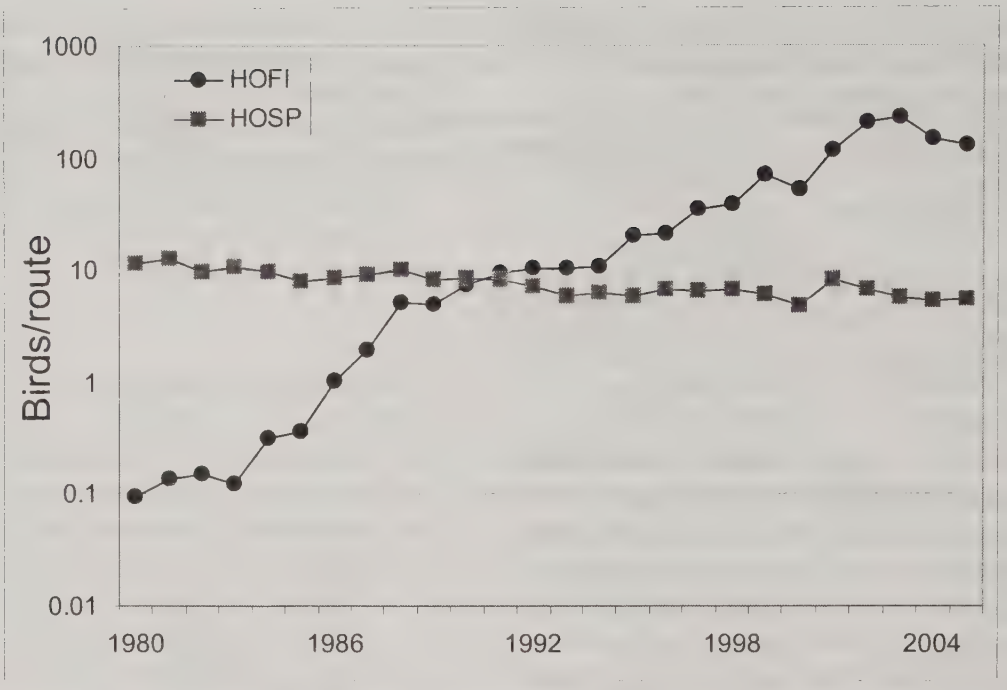



**Figure 1.** Recent trends in house finch (HOFI) and house sparrow (HOSP) abundance in Florida, Breeding Bird Survey data.

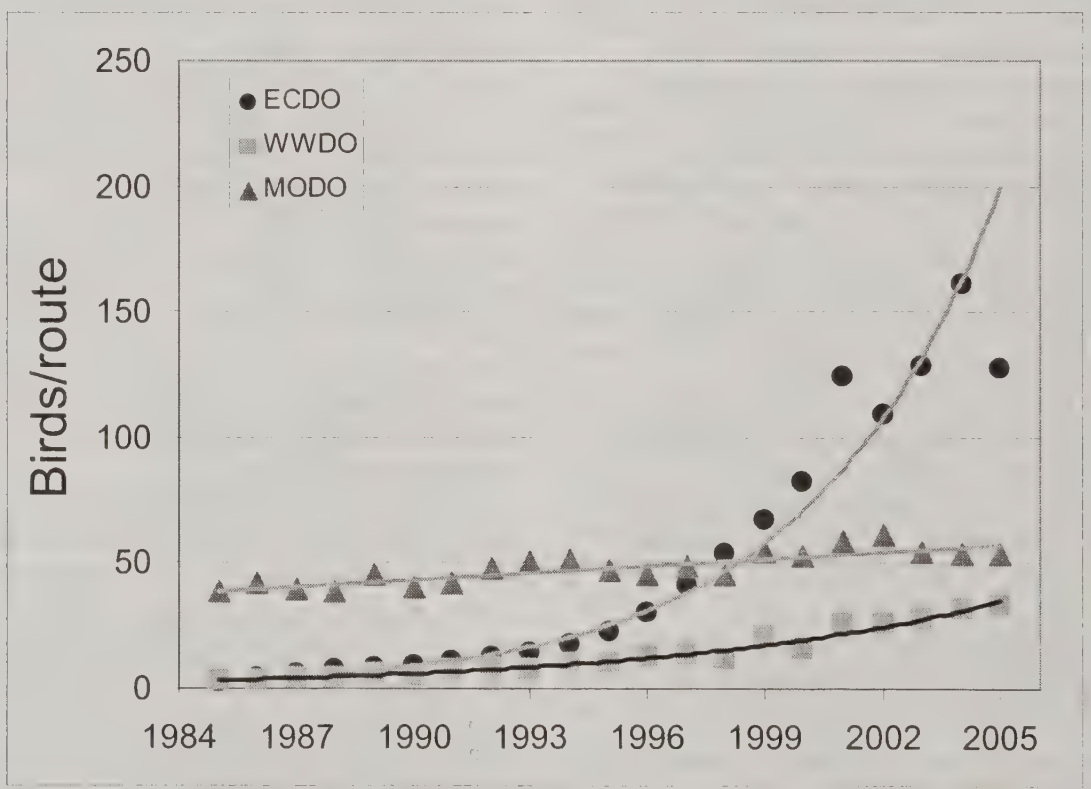



**Figure 2.** Recent trends in the abundance of Eurasian collared-dove (ECDO), white-winged dove (WWDO), and mourning dove (MODO) in Florida, Breeding Bird Survey.

exploits bird feeders and other human-derived sources of food. It appears to be in the midst of widespread and rapid range expansion and likely will continue to thrive in concert with expanding human activity (Romagosa and Labisky 2000).

*Restricted range.* There are several examples of introduced species that were once entrenched, but that subsequently declined with no overt intervention by managers or wildlife biologists (Simberloff and Gibbons 2004). In Florida, the budgerigar is a well-documented example of population boom and bust (Pranty 2001). In the 1950s and 1960s, the species was a very popular pet bird and thousands were imported annually. Invariably, some escaped or were released in west-central Florida. The birds nested in natural and man-made cavities. Many residents augmented the feral population by providing nest boxes (Pranty 2001). The Florida population peaked in the mid-1970s when several thousand budgerigars were recorded annually on the Christmas Bird Count (CBC). Throughout the late 1980s and 1990s the budgerigar population declined precipitously, and the most recent CBC yielded <50 birds (Figure 3). The reason for the demise of the feral budgerigar population in Florida is not known. Possible contributing factors include decreased availability of nest boxes, competition for nest sites, disease, and cold winter weather (Pranty 2001).

Although not nearly as numerous as the monk parakeet, the black-hooded parakeet has exhibited a strong upward population growth trend in recent years (Pranty and Lovell 2004). This species is found mainly in the central west coast of Florida where the budgerigar formerly flourished. Like the budgerigar, the black-hooded parakeet is a cavity-nesting species, so negative interactions between the 2 species over nest sites could have contributed to the decline of the budgerigar population. At this point, there is no indication that the black-hooded parakeet is having a negative impact on any native species. Nevertheless if the positive population growth trend is maintained, possible negative consequences for native cavity-nesters could result.

The spot-breasted oriole is native to Mexico and Central America, and it has been breeding in the Miami area since 1950 (Stevenson and Anderson 1994). The species remains in south Florida, possibly constrained by inability to withstand cold winters. There has been no negative impact attributed to this species.

The red-whiskered bulbul is native to China and southeast Asia and is specifically named on the Federal Injurious Species List (50CFR 16.12). It was first noted in the Kendall area of Miami-Dade

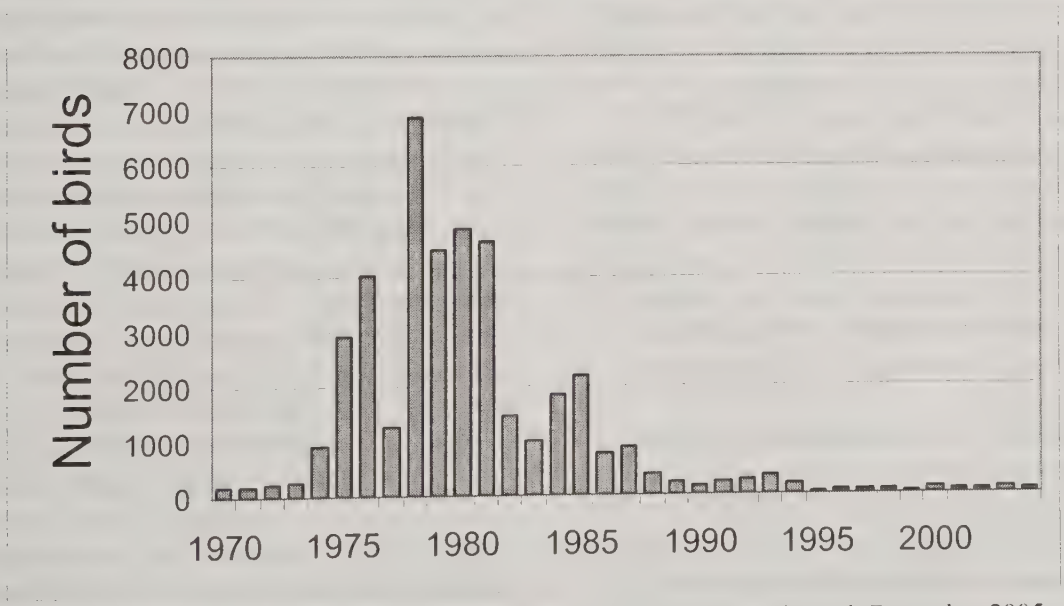



**Figure 3.** Number of budgerigars recorded on Christmas Bird Counts in Florida, through December 2005.

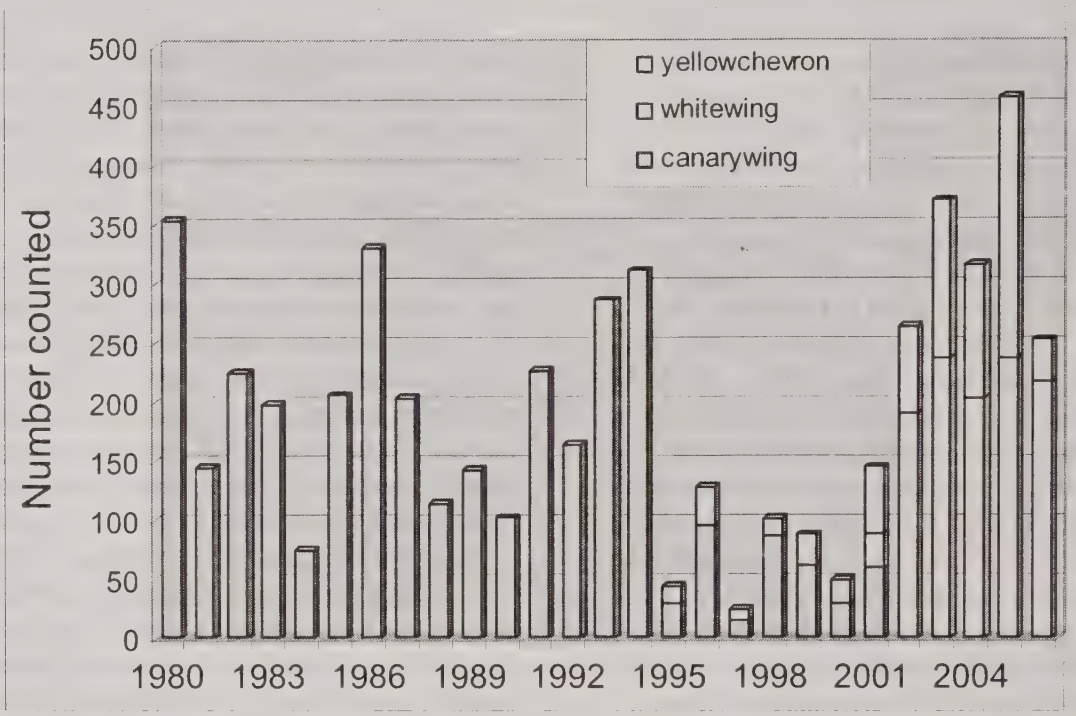



**Figure 4.** Number of *Brotegeris* parakeets recorded on Christmas Bird Counts in Florida, through December 2005.

County in the early 1960s and has persisted there where it exploits abundant exotic vegetation for nesting, roosting, and feeding (Carleton and Owre 1975). It has not expanded outside of Miami-Dade County.

The white-winged parakeet was formerly considered conspecific with the yellow-chevroned parakeet (*Brotogeris chiriri*), and, collectively, they were known as canary-winged parakeets (Brightsmith 1999). Because of possible nomenclatural confusion, and because the species are difficult to tell apart in the field, and because they are known to hybridize in Florida (Pranty and Epps 2002), we combined the *Brotogeris* data from CBC records for this paper. These records indicate that over the past 25 years there has been little net change in the size of the *Brotogeris* parakeet population in Florida other than lower numbers in 1995-2001 (Figure 4).

*Major problems*. The mallard occurs naturally in Florida in the winter, so technically it is not an exotic species. But the mallard is not a native breeding species in the state. All of the breeding seems to have been traced to domesticated birds that escaped or were released from farms, bird parks, or zoos. According to Stevenson and Anderson (1994, p. 113), "...it is possible that all summer and breeding reports have involved escaped (introduced) individuals or their descendants." Currently, mallards are breeding all over the state. The major impact of the mallard is through its propensity to breed with native species, particularly the mottled duck (*Anas fulvigula*). This genetic introgression is, in effect, causing the mottled duck to be extirpated in Florida. According to some authors, hybridization with the feral mallards "...is possibly the single greatest threat to the future of the mottled duck as a unique species" (Moorman and Gray 1994). Recognition of the problem is the first step in addressing it, but in this case it might already be too late to stop. The FWC estimates that more than 12,000 mallards are purchased statewide from feed-and-seed stores annually, and these form an ongoing pool of potential releases. It is unlawful to release mallards, but that does not stop people from continuing to release these birds on local ponds, lakes and canals for aesthetic reasons. Domesticated mallards occur year-round throughout Florida at city and county parks, apartment and condominium complexes, and in other urban and suburban areas. Feral mallards also have spread from developed areas into rural habitats.

According to the FWC, 7 to 12 percent of mottled ducks already exhibit genetic evidence of hybridization (www.myfwc.com/duck/

mottled/theproblem.htm). Because of the relatively small size of the mottled duck breeding population (estimated at 30,000-40,000), the complete hybridization could result in the extinction of the Florida mottled duck. This is a real concern because released mallards have established a track record of hybridizing and devastating local duck populations in other parts of the world. For example, in New Zealand the grey duck (*Anas superciliosa*) has become almost completely eliminated as a distinct species through hybridization with mallards released there for hunting (Braun et al. 1994). The Hawaiian duck (*Anas wyvilliana*) is a Federally-listed endangered species and genetic introgression by mallards might be the factor that seals its fate (Engilis et al. 2002). Repopulating the islands with pure-bred, captive-raised Hawaiian ducks might be possible, but only after mallards and mallard hybrids are removed.

To combat the Florida feral mallard problem, FWC is emphasizing a public information and awareness program to educate the citizens about hybridization and the laws pertaining to mallards. FWC rules stipulate that mallards cannot be possessed without a permit or released. FWC has also developed a special mallard permit program with the US Fish and Wildlife Service (USFWS) so that feral mallards can be taken during the spring and summer, outside the normal hunting season.

The monk parakeet is by far the most successful psittacine in Florida. Because of the monk parakeet's reputation as a pest species to crops in its native range in South America, early discussion of the potential impacts from the introduction and establishment of monk parakeets in the US focused on the potential of the species as a depredator of crops (Davis 1974, Neidermyer and Hickey 1977). At that time, there was no mention of the parakeet's impact on electric utility companies. Today, crop damage has not materialized as an issue except in localized situations (Tillman et al. 2001). Although crop damage by monk parakeets could still become an important issue, the most serious impact now is to electric utility companies because of power outages caused by nest-building behavior of the birds (Avery et al. 2006).

According to the CBC, the population trend for the monk parakeet was consistently upward through 2002 (Figure 5). Lately, however, the trend has been downward, but this might be only temporary. There is as yet no indication that the nests and birds removed by utility companies have negatively affected parakeet population growth.

*Emerging threats.* The common myna (*Acridotheres tristis*) is not yet on published lists of priority invasive Florida birds. Although it is so far restricted to local areas of south Florida, its numbers on the CBC have are grown steadily (Figure 6). The myna is a serious invasive problem species in other countries (Pell and Tidemann 1997, Yap et al. 2002). Although currently in Florida its limited distribution and low population size seem to pose no serious problems, there is no way to predict future events. In view of the problems caused by mynas elsewhere, it is unwise to allow the Florida population to grow unchecked.

Two recent invasive species in particular that have caught attention of Florida biologists and mangers are the purple swamphen (*Porphyrio porphyrio*) and the sacred ibis (*Threskiornis aethiopicus*). It is possible that both species escaped from the Miami Metrozoo in 1992 as a result of Hurricane Andrew. Alternatively, some might have escaped from aviculturists. Regardless, both are now showing up in natural and manmade wetlands in south Florida (Pranty et al. 2000, Herring et al 2006). It is not clear what negative consequences could result from the presence of these non-native species, but that should not be the point. While the opportunity exists to remove them from the Florida landscape, it should be done. It makes little sense to wait and study the situation to see what impacts might accrue. As management action is delayed, populations of these species will increase and spread, making it that much more difficult and expensive to implement effective corrective measures later (Simberloff 2003).

To that end, action was initiated in 2006 to limit the continued spread of the purple swamphen in south Florida. In a cooperative effort, biologists with the South Florida Water Management District (SFWMD), the USFWS, and the FWC located and removed over 800 birds during October 2006-August 2007 (Clary 2007). Retrieval efforts are scheduled to continue to remove the remainder of the introduced population.

*Formerly established.* In Florida, several bird species that once were considered to be established are now extirpated (Woolfenden et al. 2006). The black francolin (*Francolinus francolinus*) was introduced as a potential game bird to several parts of the state in the 1960s. The species persisted into the 1980s but ultimately disappeared and is now absent from the Florida avifauna (Stevenson and Anderson 1994).

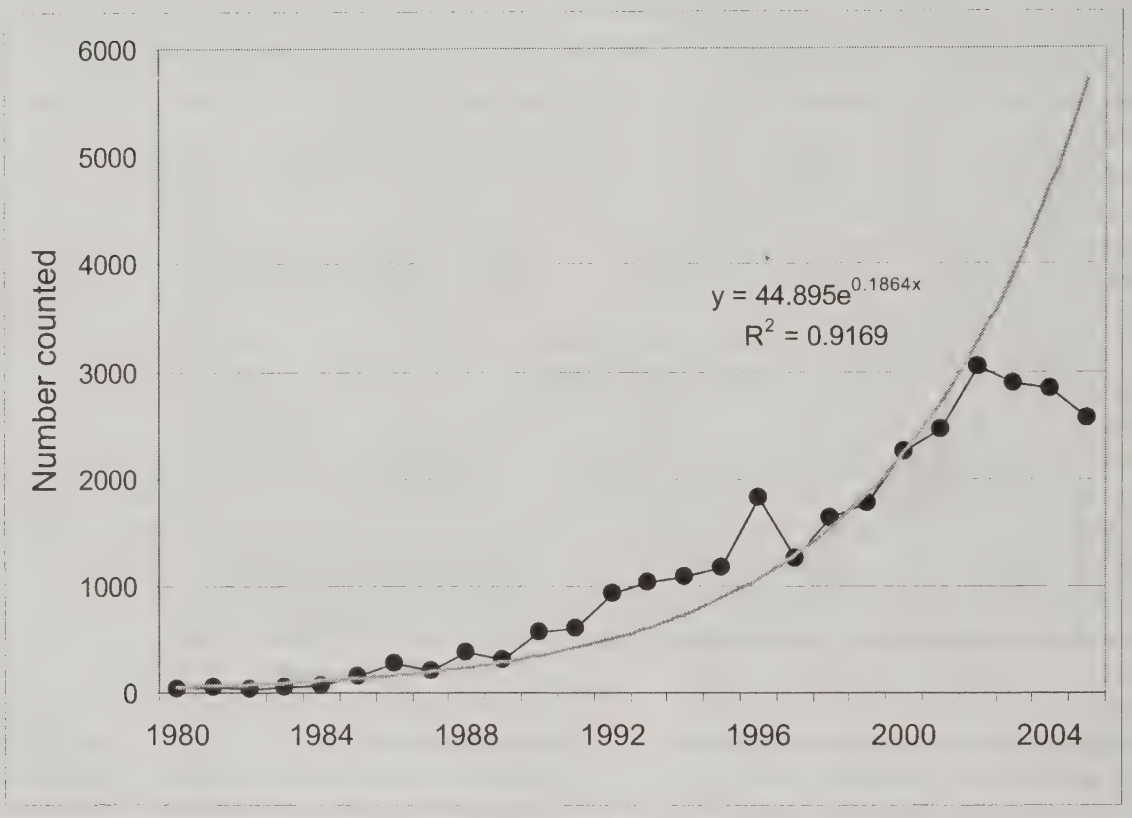



**Figure 5.** Monk parakeets recorded on the Christmas Bird Count in Florida, through December 2005.

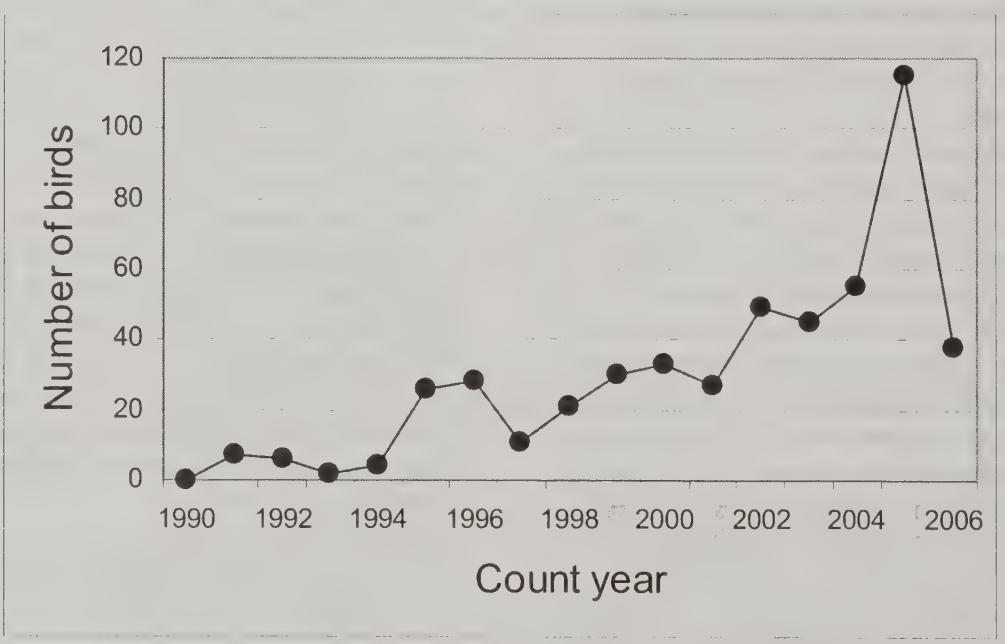



**Figure 6.** Common mynas recorded on Florida Christmas Bird Counts, through December 2005.

The blue-gray tanager (*Thraupis episcopus*), a common neotropical bird that feeds principally on fruit and nectar, was never very numerous in Florida. Throughout the 1960s and into the 1970s, however, it did breed in several locations in south Florida (Stevenson and Anderson 1994). There have been no reports of breeding birds for over 30 years, and new sightings are believed to be recently escaped individuals.

The Java sparrow (*Padda oryzivora*), a seed-eating species native to Indonesia, is successfully established in Hawaii, Puerto Rico, and many other places worldwide (Islam 1997). In Florida, breeding was reported in 3 counties, with the Miami-Dade County population seeming to thrive during the early 1970s (Stevenson and Anderson 1994). The last published report of the Java sparrow was in 1977, however, and the species is now considered extirpated.

No one has proffered explanations for the demise of these species following their initial establishment as breeding birds in Florida. None was present long enough or was sufficiently abundant to draw attention as an object for study or management. Sudden inexplicable declines or local extinctions of established invasives have been noted elsewhere (Simberloff and Gibbons 2004). This illustrates the inherent difficulty in predicting the consequences of exotic bird introductions. Alternatively, establishment of these species might have been illusory, and their seeming persistence due to repeated introductions (Moulton 1985).

**Non-native Species Trends in Florida**

The percentage of observations that are represented by non-native birds in the most recent CBC varies considerably across the state (Figure 7). There is a strong relationship between human population size and the percentage of non-native birds in the most recent CBC for 14 of the 17 most populous metro areas in Florida (Figure 8). Except for Jacksonville, coastal areas appear to have more

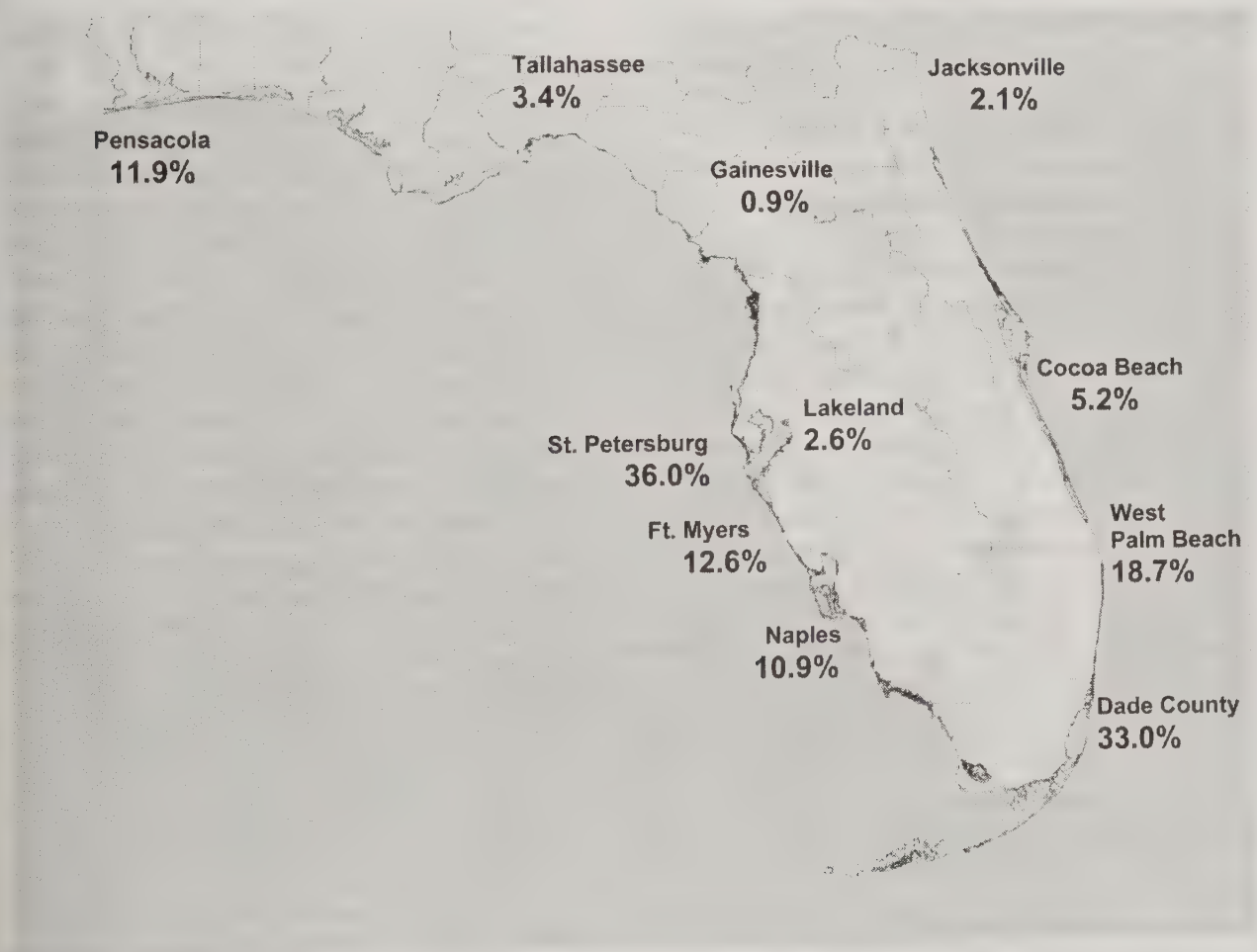



**Figure 7.** Percentage of non-native birds in Christmas Bird Count totals from 11 sites throughout Florida.

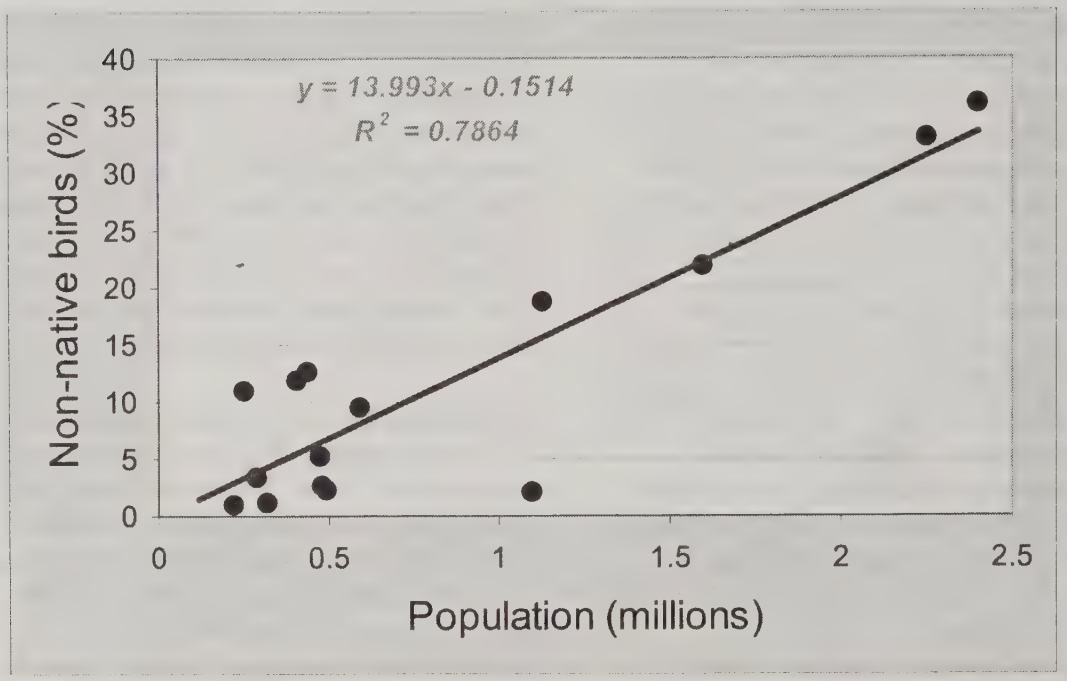



**Figure 8.** Percentage of non-native birds in total Christmas Bird Count increases with human population size among Florida's 14 largest metro areas.

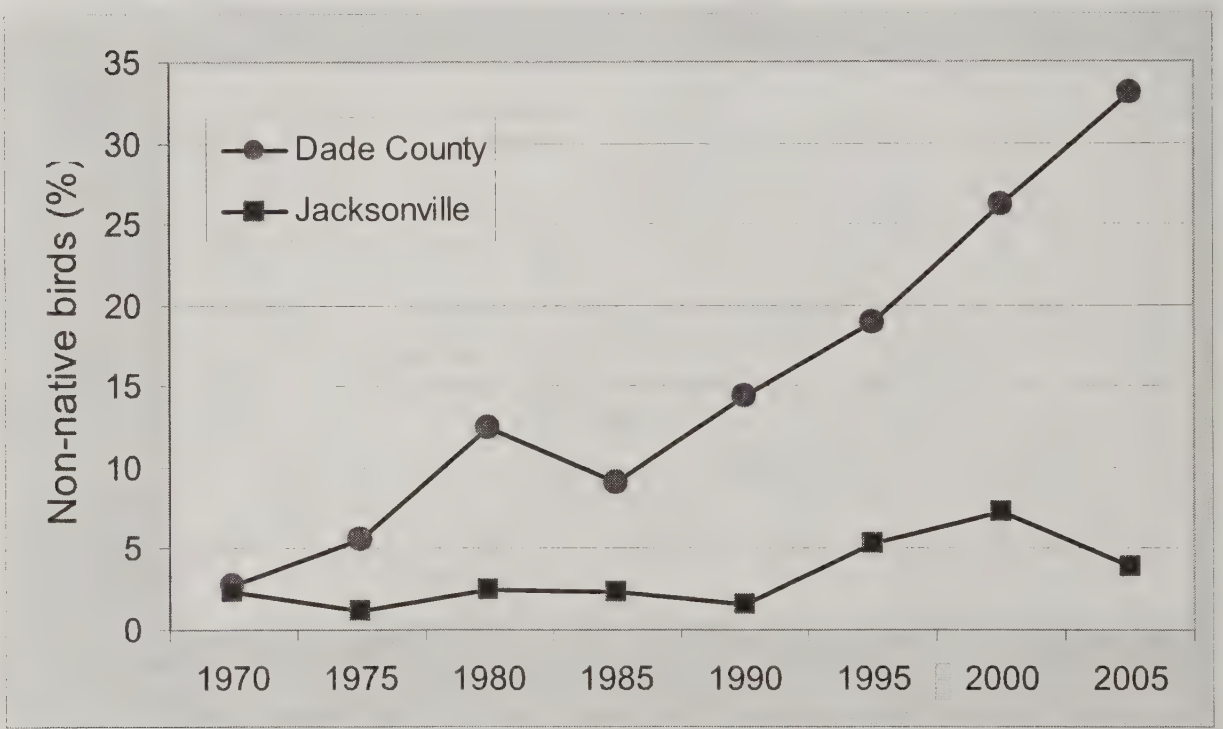



**Figure 9.** Percentage of non-native birds in total Christmas Bird Count remained virtually unchanged in Jacksonville, but increase steadily since 1970 at the Dade County site.

non-natives relative to inland sites. The high percentages of non-natives likely reflect the major conversions of native habitats that have taken place in those human population centers. Since 1970, the percentage of non-native birds in the annual CBC has remained roughly the same in Jacksonville, but it has steadily increased at the Dade County site (Figure 9).

## DISCUSSION

Birds are released, on purpose or by accident, virtually every day in countless places throughout the world. Such events usually go unnoticed and most have no lasting impact. Occasionally, however, there are repercussions. An elusive goal has been to forecast accurately the impacts of introducing a non-native species. The study of avian introductions is not an exact science. Is there such a thing as a safe introduction?

Of the approximately 200 non-native bird species recorded in Florida, only about 5% are now considered to be established. Is it possible, retrospectively, to examine the species that are established and those that are not to determine species-specific factors which would promote establishment?

A common factor among the 14 species comprising the set of Florida's established non-native birds is that each is closely tied with human disturbance and human activity. With the possible exception of the mallard, none of the 14 species in Table 1 is established and freely living in natural biotic communities independent of human support. This suggests that a key element in the successful establishment of a non-native bird species in Florida is its capacity to coexist with human activity and to exploit opportunities created by human activity. This observation is consistent with recent, more inclusive analyses that identified human affiliation and propagule pressure as the two major correlates of invasion success (Jeschke and Strayer 2006).

Not every human-affiliated bird becomes established. In Florida, more than 70 species of psittacines have been recorded, undoubtedly the result of released or escaped cage birds. Yet, just 4 of these species are on the list of established non-native birds (Table 1). To be sure, more than 4 psittacine species are breeding and might eventually become established in Florida, but obviously human affiliation is no guarantee of long-term success for a non-native bird. Jeschke and Strayer (2006) also cite propagule pressure, the number of individuals released and the number releases that occurred, as key to invasion success. For the majority of Florida species, propagule pressure is unknown. We can assume for the budgerigar that propagule pressure was quite substantial because of its popularity as a pet and the number of outdoor aviaries that existed during the 1970s (Pranty 2001). Still, this species flourished in Florida only briefly, and it currently seems destined for extirpation (Pranty 2001). For this non-native species, human affiliation and propagule pressure might have facilitated initial establishment, but were clearly insufficient for long-term population viability. We feel that species-specific traits are also critical to the process of establishment and persistence.

For the monk parakeet, one species-specific trait that no doubt improves its prospects for establishment is nest-building behavior. The monk parakeet is unique among psittacines in building a nest of sticks and branches rather than nesting in cavities. They use not only trees, but a number of man-made structures as nesting substrates (Avery et al. 2006). Monk parakeets are thus freed from the constraint of nest site availability which is certainly an advantage for a non-native species.

## MANAGEMENT IMPLICATIONS

Becoming established is an essential aspect of the invasion process, but then becoming an invasive species (i.e., one that causes economic or ecological harm) is another matter. Our knowledge of these matters is imperfect, and therefore, we are not able to predict with any certainty the outcome of non-native species establishment. The limits of our understanding of these processes are revealed by formerly established non-native bird species that for unknown reasons are no longer present where they previously flourished (Simberloff and Gibbons 2004). Rather than risk potentially serious ecological damage, we ought to adopt a rapid response mind-set that includes zero tolerance for new free-living, non-native species. To wait and see what happens when a new free-living non-native species is detected is to risk turning a manageable problem now into an intractable one later (Simberloff 2003).

The ability to respond rapidly and effectively when a non-native species is first detected presupposes the existence of personnel and resources dedicated to this effort. Also, this approach must account for adverse public reaction that can bring well-intentioned management efforts

to a screeching halt. Within a program to manage non-native species, public education efforts to raise awareness of the detrimental impacts of non-native species are crucial. Finally, in a region like South Florida, where there are numerous land management jurisdictions, cooperation among agencies will be essential for the elimination of bureaucratic barriers that unnecessarily complicate wildlife management efforts. This cooperative approach is exemplified by recent actions of the South Florida Water Management District, the USFWS, and the Florida Fish and Wildlife Conservation Commission to eradicate purple swamphen populations (Clary 2007). Birds do not respect the land management agency boundaries. Effective implementation of programs designed to control non-native birds and other wildlife entails recognition of common objectives and willingness to commit resources to achieve them.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

AVERY, M. L., J. R. LINDSAY, J. R. NEWMAN, S. PRUETT-JONES, AND E. A. TILLMAN. 2006. Reducing monk parakeet impacts to electric utility facilities in south Florida. Pages 125-136 *in* C. J. Feare and D. P.Cowan, editors. Advances in Vertebrate Pest Management, volume IV. Filander Verlag, Fürth.

BAILEY, H. H. 1928. "An ornithological enigma" explained. Auk 45:216-217.

BRAUN, M. J., RHYMER, J. M., AND WILLIAMS, M. J. 1994. Mitochondrial analysis of gene flow between New Zealand mallards (*Anas platyrhynchos*) and grey ducks (*A. superciliosa*). Auk 111:970-978.

BRIGHTSMITH, D. 1999. White-winged parakeet (*Brotogeris versicolurus*) and yellow-chevroned parakeet (*Brotogeris chiriri*). *In* A. Poole and F. Gill, editors. The birds of North America, No. 386–387. The Birds of North America, Inc., Philadelphia, Pennsylvania, USA.

CARLETON, A. R., AND O. T. OWRE. 1975. The Red-whiskered bulbul in Florida: 1960-71. Auk 92:40-57.

CLARY, M. 2007. State aims to eradicate exotic purple swamphens in wetlands. South Florida Sun-Sentinal. September 8, 2007.

COOPER, C. B., W. M. HOCHACHKA, AND A. A. DHONDT. 2007. Contrasting natural experiments confirm competition between house finches and house sparrows. Ecology 88:864-870.

DAVIS, L. R. 1974. The monk parakeet: a potential threat to agriculture. Vertebrate Pest Conference 6:253-256.

ENGILIS, A., JR., K. J. UYEHARA, AND J. G. GIFFIN. 2002. Hawaiian duck (*Anas wyvilliana*). *In* A. Poole and F. Gill, editors. The birds of North America, No. 694. The Birds of North America, Inc., Philadelphia, Pennsylvania, USA.

FARMER, K. L,, G. E. HILL, AND S. R. ROBERTS. 2005. Susceptibility of wild songbirds to the house finch strain of *Mycoplasma gallisepticum*. Journal of Wildlife Diseases 41:317-325.

HERRING, G., E. M. CALL, AND M. D. JOHNSTON. 2006. A non-indigenous wading bird breeding in the Florida Everglades: the sacred ibis. Florida Field Naturalist 34:4-8.

ISLAM, K. 1997. Java sparrow (*Padda oryzivora*). *In* A. Poole and F. Gill, editors. The birds of North America, No. 304. The Academy of Natural Sciences, Philadelphia, PA, and The American Ornithologists' Union, Washington, D.C., USA.

JESCHKE, J. M., AND D. L. STRAYER. 2006. Determinants of vertebrate invasion success in Europe and North America. Global Change Biology 12:1608-1619.

LUTTRELL, M. P., D. E. STALLKNECHT, S. H. KLEVEN, D. M. KAVANAUGH, J. L. CORN, AND J. R. FISCHER. 2001. *Mycoplasma gallisepticum* in house finches (*Carpodacus mexicanus*) and other wild birds associated with poultry production facilities. Avian Diseases 45:321-329.

MOORMAN, T. E. AND P. N. GRAY. 1994. Mottled duck (*Anas fulvigula*). *In* A. Poole and F. Gill, editors. The birds of North America, No. 81. Philadelphia: The Academy of Natural Sciences; Washington, D.C.: The American Ornithologists' Union, USA.

MOULTON, M. P. 1985. Morphological similarity and the coexistence of congeners: an experimental test using introduced Hawaiian birds. Oikos 44:301-305.

NEIDERMYER, W. J., AND J. J. HICKEY. 1977. The monk parakeet in the United States, 1970–1975. American Birds 31:273–278.

OWRE, O. T. 1973. A consideration of the exotic avifauna of southeastern Florida. Wilson Bulletin 85:491-500.

PELL, A. S., AND C. R. TIDEMANN. 1997. The impact of two exotic hollow-nesting birds on two native parrots in savannah and woodland in eastern Australia. Biological Conservation 79:145-153.

POLING, T. D., AND S. E. HAYSLETTE. 2006. Dietary overlap and foraging competition between mourning doves and Eurasian collared-doves. Journal of Wildlife Management 70:996-1004.

PRANTY, B. 2001. The budgerigar in Florida: rise and fall of an exotic psittacid. North American Birds 55:389-397.

PRANTY, B., AND S. EPPS. 2002. Distribution, population status, and documentation of exotic parrots in

Broward County, Florida. Florida Field Naturalist 30:111-131.

PRANTY, B., AND H. W. LOVELL. 2004. Population increase and range expansion of black-hooded parakeets in Florida. Florida Field Naturalist 32:129-137.

PRANTY, B., SCHNITZIUS, K., SCHNITZIUS, K., AND LOVELL, H. W. 2000. Discovery, origin, and current distribution of the purple swamphen (*Porphyrio porphyrio*) in Florida. Florida Field Naturalist 28:1-11.

ROBINSON, R. A., G. M. SIRIWARDENA, AND H. Q. P. CRICK. 2005. Size and trends of the house sparrow *Passer domesticus* population in Great Britain. Ibis 147:552-562.

ROMAGOSA, C. M., AND R. F. LABISKY. 2000. Establishment and dispersal of the Eurasian collared-dove in Florida. Journal of Field Ornithology 71:159-166.

SAUER, J. R., J. E. HINES, AND J. FALLON. 2005. The North American breeding bird survey, results and analysis 1966 - 2005. Version 6.2.2006. USGS Patuxent Wildlife Research Center, Laurel, Maryland, USA.

SIMBERLOFF, D. 2003. How much information on population biology is needed to manage introduced species? Conservation Biology 17:83-92.

SIMBERLOFF, D., AND L. GIBBONS. 2004. Now you see them, now you don't! – population crashes of established introduced species. Biological Invasions 6:161-172.

STEVENSON, H. M., AND B. H ANDERSON. 1994. The birdlife of Florida. University Press of Florida. Gainesville, Florida, USA.

TILLMAN, E. A., A. VAN DOORN, AND M. L. AVERY. 2001. Bird damage to tropical fruit in south Florida. Proceedings of the Wildlife Damage Management Conference 9:47-59.

WILD BIRD CONSERVATION ACT. 1992. Pub. L. 102-440, 106 Stat. 2224, October 23, 1992.

WISE, J. K., B. L. HEATHCOTT, AND M. L. GONZALEZ. 2002. Results of the AVMA survey on companion animal ownership in US pet-owning households. Journal of the American Veterinary Medical Association 221:1572-1573.

WOOLFENDEN, G. E., W. B. ROBERTSON, JR., AND J. A. COX. 2006. The breeding birds of Florida. Florida Ornithological Society Special Publication 7.

YAP, C. A.-M., N. S. SODHI, AND B. W. BROOK. 2002. Roost characteristics of invasive mynas in Singapore. Journal of Wildlife Management 66:1118-1127.

# EUROPEAN STARLINGS: A REVIEW OF AN INVASIVE SPECIES WITH FAR-REACHING IMPACTS

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***Abstract:*** The introduction of European starlings (*Sturnus vulgaris*) in New York City in 1890 and 1891 resulted in their permanent establishment in North America. The successful occupation of North America (and most other continents as well) has earned the starling a nomination in the Top 100 list of 'Worlds Worst' invaders. Pimentel et al. (2000) estimated that starling damage to agriculture crops in the United States was $800 million yearly, based on $5/ha damage. Starlings may spread infectious diseases that sicken humans and livestock, costing nearly $800 million in health treatment costs. Lastly, starlings perhaps have contributed to the decline of native cavity-nesting birds by usurping their nesting sites. We describe the life history of starlings, their economic impact on agriculture, and their potential role as vectors in spreading diseases to livestock and humans. We recommend that the database on migratory and local movements of starlings be augmented and that improved baits and baiting strategies be developed to reduce nuisance populations.

***Key Words:*** agricultural damage, disease, European starling, invasive species, livestock, *Sturnus vulgaris*, urban wildlife.

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## INTRODUCTION

The European starling (*Sturnus vulgaris*) has been nominated by the Invasive Species Specialist Group to the "100 World's Worst" invaders (Lowe et al. 2004). Only two other bird species, the common myna (*Acridotheres tristis*) and red-vented bulbul (*Pycnonotus cafer*), have been nominated for this list. In this paper, we review the biology and ecology behind the European starling's phenomenal adaptive abilities. We also describe case studies of damage caused by starlings and overall economic impact. We focus on the European starlings (henceforth, starlings) in North America, but acknowledge that starlings have been introduced into or invaded most other continents as well, and likewise have become a successfully established pest species around the world.

## THE INVASION

The European starling's native range is Europe, southwest Asia and northern Africa. It was introduced into North America, South Africa, New Zealand, and Australia because of cultural longings of new immigrants and false perceptions of biological control of insects (Adency 2001). Becoming a viable invading population is not always easy, even for a species as highly adapted to human environments as the starling. In North America, many attempts were made at introduction. In the mid- to late nineteenth century, people in both Oregon and New York tried repeatedly. Starlings eventually took hold in New York City, where 16 pairs survived a 50-pair release in 1890 and 1891. The descendants of these survivors expanded their population rapidly westward, reaching the West Coast in 1942 (see Cabe 1993, for an excellent review of this species). It is

believed that most or all of the starlings now inhabiting North America (from Florida to Alaska), the Bahamas, Central America, Yucatan Peninsula, Puerto Rico, Jamaica, and Cuba are from the original 16 pairs that colonized New York City. North America's population is estimated at 200 million, about one-third of the world's starling population (Feare 1984). According to route counts from the North American Breeding Bird Survey, only red-winged blackbirds (*Agelaius phoeniceus*) out-number starlings (US Geological Survey 2005).

## IDENTIFICATION

In breeding plumage, this small bird (70-100 g) has iridescent green feathers that cover the nape, breast and back; whereas, their pointed, triangular, black wings (31-40 cm long) have a green or purple sheen. The bill color helps distinguish the sexes during the breeding season. Both genders have a bright yellow beak; however, the lower mandible is blue-gray in males and pink in females during the breeding season. Uniform brown eyes are typical of males; the female has a narrow and light-colored iris. Mass of males (73-96 g) and females (69-93 g) overlap to the point that this metric is unreliable for distinguishing between genders (Blem 1981).

## REPRODUCTION AND CARE OF YOUNG

Once established at a site, starlings have a high degree of breeding site fidelity. Kessel (1957) found that about 30% of females used the same nest box in successive years, and about 90% moved less than 1 km to breed. The young-of-the-year disperse widely and find new breeding sites, often far away from their natal site. Eggs are laid from late March to early July, depending on latitude. Nest sites are in natural cavities or in various nooks and crannies of man-made structures. Most starlings produce 1 to 2 clutches per year of 4-6 eggs each, with birds above $48^0$ N producing only one clutch (Craig and Feare 1999). Eggs are incubated for about 12 days before hatching (Ricklefs and Smeraski 1983). Experiments have shown that if the first egg is removed, females sometimes continue laying in an attempt to complete the clutch (Meijer 1990); at other times, they do not replace the removed first egg (Kennedy and Power 1990). The incubation period lasts for about 12 days. The eggs are incubated for 18-19 hours per day, and up to 23 hours per day during cold periods. The female does about 70% of the incubation during the day and all of it at night (Drent et al. 1985). As a result of nest parasitism, up to 33% of the nestlings are not genetic siblings (Pinxten et al. 1987, Power et al. 1989, Romagnano et al. 1990). Parents bring food to the nest up to 20 times per hour. Nestlings show fear by day 12, and thermoregulate by day 13 (Johnson and Cowan 1974). It takes about 3 weeks for nestlings to fledge. The parents continue feeding the fledglings for another 4-10 days before complete independence is achieved. The average life span is about 2-3 years, with a longevity record of over 20 years.

## FOODS

Starlings eat plant matter and invertebrates, with the latter being favored during the spring and summer months, when they are readily available. Starlings feed on a wide variety of invertebrates, including beetles, millipedes, butterfly and moth larvae, grasshoppers, and crickets (Tinbergen 1981). Starlings forage in fruit orchards, especially in the fall. In the winter, particularly when the ground is snow-covered or frozen, they frequent feedlots, dairies, and urban landfills, where food is abundant. Starlings often join large roosting aggregations that exploit a nutritious and abundant food source (Morrison and Caccamise 1990, Caccamise 1991). Once established at a preferred feeding site, they may be hard to dissuade from it and will endure tremendous efforts by humans at keeping them away. In caged trials, researchers found that starlings eat 7-23 g of animal food daily and 20-40 g of plant food (Feare 1984). Fischl and Caccamise (1987) found that plant food made up 62% of starlings' diet by dry weight; 21% was animal matter, but this ratio varies seasonally.

## POPULATION DYNAMICS AND MORTALITY FACTORS

Starlings are prolific and have a 48% to 79% rate of nest success (Kessel 1957, Royall 1966). Even so, only 20% nestlings survive to reproduce (Kessel 1957). Adult survival is much higher, probably around 60% (Flux and Flux 1981). It is not known what is limiting the starling population in North America. Starlings carry a heavy parasite load, including lice, mites and ticks, flatworms and round worms, all of which affect mortality rates (Boyd 1951). Cold and wet weather and extreme hot weather can contribute to mortality of nestlings, with both factors affecting the availability of an important food source, temperature sensitive invertebrates (Gromadzki 1980, Tinbergen 1981).

Occasionally, squirrels access nests and destroy entire clutches (Feare 1984). Finally, large numbers of starlings are trapped and poisoned to protect agriculture, livestock, and urban and industrial structures, but this probably has little impact on the overall population (Homan et al. 2005). Availability of nesting sites may limit the populations.

## MIGRATION

Starlings are strong flyers and can, if necessary, migrate distances of 1,000-1,500 km, especially to escape heavy snow that covers food sources. They can migrate long distances in a single day at speeds of 60-80 km/h (Feare 1984), stopping to forage along the way. Migration patterns vary by year, by region, and by individual (Kessel 1953, Suthers 1978). Starlings south of 40° N rarely migrate (Dolbeer 1982). Spring migration occurs from mid-February to late March and fall migration from September to early December (Kessel 1953, Dolbeer 1982).

## DISEASE TRANSMISSION

Starlings carry a plethora of diseases (Weber 1979, Gautsch et al. 2000, Clark and McLean 2003, Table 1). Avian salmonellosis (primarily, *Salmonella enterica*) has been documented in starlings (Feare 1984). This disease is transmissible to humans, poultry, and livestock. Chlamydiosis (also psittacosis, ornithosis, parrot fever) usually results from inhaling *Chlamydophila psittaci* that lives in dried feces. Starlings and blackbirds can infect humans and domestic fowl with *C. psittaci* (Grimes 1978, Grimes et al. 1979, Andersen et al. 1997). Starlings also carry *Mycobacterium avium paratuberculosis*, which causes Johne's disease in cattle (also known as paratuberculosis) (Matthews and McDiarmid 1979, Corn et al. 2005). The bacteria are excreted in feces and milk. Johne's disease costs the United States (US) dairy industry $200-250 million, annually (Beard et al. 2001, Ott et al. 1999). Starling fecal matter can pass transmissible gastroenteritis (TGE) to swine. Although the evidence is largely indirect and circumstantial, it is believed that during the winter of 1978-1979 starlings served as vectors for an outbreak of TGE in Nebraska that caused the loss of 10,000 swine in one month (Pilchard 1965, Bohl 1975, Gough et al. 1979, Johnson and Glahn 1994). Shiga toxin-producing *Escherichia coli* (STEC) is another disease the may be transmitted by starlings to cattle. In the cattle industry, annual costs of illnesses related to E. coli STEC exceeded $267 million (NCBA 2004). Humans get this disease when consuming tainted food products, especially ground beef. Knowledge of the movement patterns of starlings would be critical to understanding the real role that starlings have in epidemiologies of these diseases.

By disturbing soil or flooring at blackbird and starling roosts, humans can become ill with histoplasmosis, a fungal disease of the lungs caused by *Histoplasma capsulatum* (DiSalvo and Johnson 1979, Storch et al. 1980). Histoplasmosis recently was reported at a manufacturing facility in Nebraska used by starlings (J. Hobbs, personal communication). People at highest risk of exposure, however, are those working in agriculture, particularly poultry, or those coming in contact with bird or bat roosts that might have been abandoned a decade or more prior to disturbance (DiSalvo and Johnson 1979). Finally, West Nile virus (WNV) was confirmed in North America in 1999 and since that time has spread across the US. This is a serious, and life-threatening disease to humans and wildlife. Sullivan et al. (2006) found that red-winged blackbirds are WNV hosts and can disperse diseases along their migratory routes. The role of starlings in dispersing WNV is unknown, but starlings can act as hosts for the virus (Bernard et al. 2001), and thus may be involved in spreading the disease among vertebrates including, humans, horses, and birds.

## ECONOMIC IMPACTS

Pimentel et al. (2000) estimated that yearly starling damage to agriculture was US$800 million, based on a figure of US$5/ha. In 1999, three feedlot operators in Kansas estimated a loss of $600,000 from bird damage alone (US Department of Agriculture 2000). Data reported in 1968 from Colorado feedlots indicated the cost of cattle rations consumed during winter by starlings was $84 per 1,000 starlings. With the current cost of feed, the associated losses would certainly be much higher. In Idaho, some livestock facility operators estimated that starlings consumed 15 to 20 tons of cattle feed per day. The costs associated with starlings in the spread of livestock disease may be more important than food consumption. For example, the 10,000 pigs lost in Nebraska might be valued at nearly US$1.0 million in today's market.

**Table 1.** Information on some diseases transmissible to humans and livestock that are associated with feral domestic pigeons, European starlings, and English sparrows. Data originally from Weber (1979) and accessed in numerous Wildlife Services' Environmental Assessments.

| Disease | Livestock Affected | Symptoms | Comments |
|---|---|---|---|
| Bacterial: | | | |
| Erysipeloid | Cattle, swine, horses, sheep, goats, chickens, turkeys, ducks | Pigs - arthritis, skin lesions, necrosis, septicemia<br>Sheep - lameness | Serious hazard for the swine industry, rejection of swine meat at slaughter due to speticemia, also affects dogs |
| Salmonellosis | All domestic animals | Abortions in mature cattle, mortality in calves, decrease in milk production in dairy cattle<br>Colitis in pigs, | Over 1,700 serotypes |
| Pasteurellosis | Cattle, swine, horses, rabbits, chickens, turkeys | Chickens and turkeys die suddenly without illness pneumonia, bovine mastitis, abortions in swine, septicemia, abscesses | Also affects cats and dogs |
| Avian tuberculosis | Chickens, turkeys, swine, cattle, horses, sheep | Emaciation, decrease in egg production, and death in poultry. Mastitis in cattle | Also affects dogs and cats |
| Streptococcosis | Cattle, swine, sheep, horses, chickens, turkeys, geese, ducks, rabbits | Emaciation and death in poultry. Mastitis in cattle, abscesses and inflamation of the heart, and death in swine | Feral pigeons are susceptible and aid in transmission |
| Yersinosis | Cattle, sheep, goats, horses, turkeys, chickens, ducks | Abortion in sheep and cattle | Also affects dogs and cats |
| Vibriosis | Cattle and sheep | In cattle, often a cause of infertility or early embryonic death. In sheep, the only known cause of infectious abortion in late pregnancy | Of great economic importance |

**Table 1.** Continued

| Disease | Livestock Affected | Symptoms | Comments |
|---|---|---|---|
| Bacterial (continued): | | | |
| Listeriosis | Chickens, ducks, geese, cattle, horses, swine, sheep, goats | In cattle, sheep, and goats, difficulty swallowing, nasal discharge, paralysis of throat and facial muscles | Also affects cats and dogs |
| Viral: | | | |
| Meningitis | Cattle, sheep, swine, poultry | Inflammation of the brain, newborn calves unable to suckle | Associated with listeriosis, salmonellosis, cryptococcosis |
| Encephalitis (7 forms) | Horses, turkeys, ducks | Drowsiness, inflammation of the brain | Mosquitos serve as vectors |
| Mycotic (fungal): | | | |
| Aspergillosis | Cattle, chickens, turkeys, ducks | Abortions in cattle | Common in turkey poults |
| Blastomycosis | Weight loss, fever, cough, bloody sputum, chest pains. | Rarely | Affects horses, dogs and cats |
| Candidiasis | Cattle, swine, sheep, horses, chickens, turkeys | In cattle, mastitis, diarrhea, vaginal discharge, and aborted fetuses | Causes unsatisfactory growth in chickens |
| Cryptococcosis | Cattle, swine, horses | Chronic mastitis in cattle, decreased milk flow and appetite loss | Also affects dogs and cats |
| Histoplasmosis | Horses, cattle, swine | Chronic cough (in dogs), loss of appetite, weakness, depression, diarrhea, extreme weight loss | Also affects dogs; actively grows and multiplies in soil and remains active long after birds have departed |
| Coccidiosis | Poultry, cattle, and sheep | Bloody diarrhea in chickens, dehydration, retardation of growth | Almost always present in English sparrows; also found in pigeons and European starlings |

**Table 1.** Continued.

| Disease | Livestock Affected | Symptoms | Comments |
|---|---|---|---|
| Protozoal: | | | |
| American trypanosomiasis | Infection of mucous membranes of eyes or nose, swelling | Possible death in 2-4 weeks | Caused by the conenose bug found on pigeons |
| Toxoplasmosis | Cattle, swine, horses, sheep, chickens, turkeys | In cattle, muscular tremors, coughing, sneezing, nasal discharge, frothing at the mouth, prostration and abortion | Also affects dogs and cats |
| Rickettsial/Chlamydial: | | | |
| Chlamydiosis | Cattle, horses, swine, sheep, goats, chickens, turkeys, ducks, geese | In cattle, abortion, arthritis, conjunctivitis, enteritis | Also affects dogs, cats, and many wild birds and mammals |
| Q fever | Affects cattle, sheep, goats, and poultry | May cause abortions in sheep and goats | Can be transmitted by infected ticks |

Starlings also damage fruit and grain crops (Johnson and Glahn 1994). Bird damage to grapes in the US was estimated to be at least $4.4 million in 1972; and starlings were one of the species that caused the most damage. Starlings also damage ripening cherry crops. A 1972 study in Michigan found 17% of a total crop was lost. Starlings damage other cultivated fruit crops, such as peaches, blueberries, strawberries, figs, and apples. Although not a major consumer of cereal grains, starlings have damaged winter wheat and ripening corn. In Kentucky and Tennessee, starling damage to winter wheat averaged 1.6%, with higher losses occurring near roosts.

Starling roosts near airports pose an aircraft safety hazard because of the potential for birds to be ingested into jet engines, resulting in aircraft damage or loss and, at times, injury and loss of human life. Starlings have caused some of the most disastrous bird-aircraft strikes due to their body density and flocking behavior. Starlings caused an Electra aircraft to crash in Boston in 1960 that killed 62 people. Barras et al. (2003) reported that during 1990-2001, 852 strikes involving blackbirds and starlings were reported to the FAA from 46 States and the District of Columbia. Damage was reported in 39 strikes and damage was unknown in 215 strikes; costs totaled $1,607,317. Although only about 6% of bird-aircraft strikes are associated with starlings or blackbirds, these species represent a substantial management challenge at airports.

The use of urban areas by wintering flocks of starlings seeking warmth and shelter for roosting can have serious consequences. Large roosts in buildings and industrial structures cause filth, noise, odor, and health and safety hazards. Additionally, the droppings are corrosive to infrastructure. In the early 2000s, a large population in Omaha used a bank building as a roost, costing $200,000 in cleanup costs. Nebraska Wildlife Service's (WS) provided technical training for executing an effective starling harassment strategy that significantly lowered the roosting population at the bank. Moreover, WS augmented this effort by managing the Omaha starling population with DRC-1339 avicide. Similarly, downtown Indianapolis had a problem with roosting starlings during winter. Thousands of starlings were using

the downtown area. City managers funded WS to conduct a successful comprehensive harassment program (J. Loven, personal communication). To date, population management has not been deemed necessary by the citizens of Indianapolis.

## ECOLOGICAL IMPACTS

Starlings compete aggressively with native cavity-nesting birds for nest sites. However, Koenig (2003) used Christmas Bird Counts and Breeding Bird Surveys and found the effects of starlings on populations of 27 cavity-nesting species to be practically nonexistent. He did conclude that sapsuckers (*Sphyrapicus* spp.) may have declined because of starling competition for nest sites.

## MANAGEMENT

The most universal technique of managing nuisance populations of starlings is by harassing them with propane exploders, pyrotechnics, hawk kites, and ultrasonic sounds. Unfortunately, the results from these nonlethal techniques are usually temporary, unless they are intense, sustained, and varied. In addition to physical frightening agents, the chemical frightening agent, 4-aminopyridine (Avitrol®) is sometimes used. It is a restricted use pesticide available only to certified applicators. Avitrol® baits contain a small number of treated grains or pellets that are diluted with untreated grains or pellets. Birds that eat the treated baits behave erratically and give warning cries that frighten other birds from the area. Any bird, target or non-target, that eats a 4-aminopyridine treated particle could die. In theory, hawks and owls that eat affected or dead birds that have ingested this agent also could die. Polybutenes, formulated under trade names such as Roost-No-More®, Bird Tanglefoot®, and 4-The-Birds®, are sticky materials that might discourage starlings from roosting on ledges and beams. Labor-costs and longevity are issues that managers must consider when using these nonlethal products. Finally, dimethyl anthranilate (DMA) and methyl anthranilate (MA) are taste aversive compounds that repel starlings under experimental conditions in feedlots and fruit production facilities, but apparently are not widely used for that purpose (Glahn et al. 1989). Trapping is common in some areas, followed by euthanasia.

Lethal chemical control of starling populations is achieved with DRC-1339 (3-chloro-4-methylaniline hydrochloride, also 3-chloro p-toluidine hydrochloride, 3-chloro-4-methylaniline). This avicide was first used to control starlings and blackbirds in livestock feedlots of the intermountain region of the western US (DeCino et al. 1966, Royall et al. 1967, West 1968). According to Besser et al. (1967), a starling population was reduced by about 75% after spreading 1% DRC-1339-treated poultry pellets at a cattle feedlot in Nevada. A roost of approximately 250,000 starlings in Colorado was reduced by more than 60% by baiting a feedlot and a pasture (West 1968).

Starlicide, a DRC-1339 product, is a slow-acting toxicant for controlling starlings and blackbirds around livestock and poultry operations. It is toxic to other types of birds in differing amounts, but will not kill many members of the family Emberizidae at registered levels. Mammals are generally resistant to its toxic effects. Poisoned birds experience a slow, nonviolent death. They usually die from 1 to 3 days after feeding, often at their roost. Generally, few dead starlings will be found at the bait site. Poisoned starlings are not dangerous to scavengers or predators as the chemical is quickly metabolized and excreted (Eisemann et al. 2003).

## RESEARCH NEEDS

The reproductive biology of starlings is well documented, but data on regional migratory patterns and local movements in relation to feedlots, diaries, and urban areas are needed. Scientists for USDA Wildlife Services, in collaboration with North Dakota State University, The Ohio State University, and others are beginning to gather these data. This information will be useful for developing risk assessments and economic impact models that will help determine the overall consequences of a burgeoning population of starlings. Finally, efforts are underway to develop and evaluate better bait carriers for the compound DRC-1339 baits. Finally, better information is needed to determine their role in transmitting diseases.

## LITERATURE CITED

ADENEY, J. 2001. European starling (*Sturnus vulgaris*), Columbia University's Introduced Species Summary Project website entry, accessed 22 June 2007.

ANDERSEN, A. A., J. E. GRIMES, P. B. WYRICK, B. W. CALNEK, H. J. BARNES, C. W. BEARD, L. R. MCDOUGALD, AND Y. M. SAIF. 1997. Chlamydiosis. Pages 333-349 *in* B. W. Calnek,

editor. Diseases of poultry. Tenth edition. Iowa State University Press, Ames, Iowa, USA.

Barras, S. C., S. E. Wright, and T. W. Seamans. 2003. Blackbird and starling strikes to civil aircraft in the United States, 1990-2001. Pages 91-96 *in* G. M. Linz, editor. Proceedings of symposium on management of North American blackbirds. U.S. Department of Agriculture, Animal and Plant Health Inspection Service, Wildlife Services, National Wildlife Research Center, Fort Collins, Colorado, USA.

Beard, P. M., M. J. Daniels, D. Henderson, A. Pirie, K. Rudge, D. Buxton, S. Rhind, A. Greig, M. R. Hutchings, I. McKendric, K. Stevenson, and J. M. Sharp. 2001. Paratuberculosis infection of nonruminant wildlife in Scotland. Journal of Clinical Microbiology 39:1517-1521.

Bernard, K. A., J. G. Maffei, S. A. Jones, E. B. Kauffman, G. Ebel, and A. P. Dupuis. 2001. West Nile virus infection in birds and mosquitoes, New York State 2000. Emerging Infectious Diseases 7:679-685.

Besser, J. F., W. C. Royall, Jr., and J. W. DeGrazio. 1967. Baiting starlings with DRC-1339 at a cattle feedlot. Journal of Wildlife Management 31:48-51.

Blem, C. R. 1981. Geographic variation in mid-winter body composition of starlings. Condor 83:370–376.

Bohl, E. H. 1975. Transmissible gastroenteritis. Pages168-188 *in* H.W. Dunne and A.D. Leman, editors. Diseases of swine. Fourth edition. Iowa State University Press, Ames, Iowa, USA.

Boyd, E. M. 1951. A survey of parasitism of the starling *Sturnus vulgaris* L. in North America. Journal of Parasitology 37:56–84.

Cabe, P. R. 1993. European starling (*Sturnus vulgaris*). *In* The Birds of North America, No. 48. A. Poole and F. Gill, editors. The Academy of Natural Sciences, Philadelphia, Pennsylvania and The American Ornithologists' Union, Washington, D.C, USA.

Caccamise, D. F. 1991. European starling fidelity to diurnal activity centers: role of foraging substrate quality. Wilson Bulletin 103:13–24.

Clark, L., and R. G. McLean. 2003. A review of pathogens of agricultural and human health interest found in blackbirds. Pages 103-108 *in* G. M. Linz, editor. Proceedings of symposium on management of North American blackbirds. U.S. Department of Agriculture, Animal and Plant Health Inspection Service, Wildlife Services, National Wildlife Research Center, Fort Collins, Colorado, USA.

Corn, J. L., E. J. B. Manning, S. Sreevatsan, and J. R. Fischer. 2005. Isolation of *Mycobacterium avium* subsp. *paratuberculosis* from free-ranging birds and mammals on livestock premises. Applied and Environmental Microbiology 71:6963–6967.

Craig, A., and C. Feare. 1999. The Starling. Princeton University Press, Princeton, New Jersey, USA.

DeCino, T. J., D. J. Cunningham, and E. W. Schafer, Jr. 1966. Toxicity of DRC-1339 to starlings. Journal of Wildlife Management 30:249-253.

Disalvo, A. F., and W. M. Johnson. 1979. Histoplasmosis in South Carolina: support for the microfocus concept. American Journal of Epidemiology 109:480-492.

Dolbeer, R. A. 1982. Migration patterns for age and sex classes of blackbirds and starlings. Journal of Field Ornithology 53:28–46.

Drent, R. H., J. M. Tinbergen, and H. Biebach. 1985. Incubation in the starling (*Sturnus vulgaris*): resolution of the conflict between egg care and foraging. Netherlands Journal of Zoology 35:103–123.

Eisemann, J. D., P. A. Pipas, and J. L. Cummings. 2003. Acute and chronic toxicity of compound DRC-1339 (3-chloro-4-methylaniline hydrochloride) to birds. Pages 24-28 *in* G. M. Linz, editor. Proceedings of symposium on management of North American blackbirds. U.S. Department of Agriculture, Animal and Plant Health Inspection Service, Wildlife Services, National Wildlife Research Center, Fort Collins, Colorado, USA.

Feare, C. J. 1984. The starling. Oxford University Press, New York, New York, USA.

Fischl, J., and D. F. Caccamise. 1987. Relationships of diet and roosting behavior in the European starling. American Midland Naturalist 117:395–404.

Flux, J. E. C., and M. M. Flux. 1981. Population dynamics and age structure of starlings (*Sturnus vulgaris*) in New Zealand. New Zealand Journal of Ecology 4:65–72.

Gautsch, S., P. Odermatt, A. P. Burnens, J. Bille, and R. Ewald (Translated abstract). 2000. The role of common starlings (*Sturnus vulgaris*) in the epidemiology of bacterial, potentially human pathogenic, disease agents. Schwiez Arch Tierheilkd 142:165-72.

Glahn, J. F., J. R. Mason, and D. R. Woods. 1989. Dimethyl anthranilate as a bird repellent in livestock feed. Wildlife Society Bulletin 17:313-320.

Gough, P. M., J. W. Beyer, and R. D. Jorgenson. 1979. Public health problems: TGE. Proceedings of the Bird Control Seminar 8:136-141.

Grimes, J. E. 1978. Transmission of chlamydiae from grackles to turkeys. Avian Diseases 22:308-312.

Grimes, J. E., K. J. Owens, and J. R. Singer. 1979. Experimental transmission of chlamydia psittaci to turkeys from wild birds. Avian Diseases 23:915-926.

Gromadzki, M. 1980. Reproduction of the starling *Sturnus vulgaris* in Zulawy Wislane, North Poland. Acta Ornithologica 17:195–223.

HOMAN, H. J., R. S. STAHL, J. J. JOHNSTON, AND G. M. LINZ. 2005. Estimating DRC-1339 mortality using avian bioenergetics: a case study of the European starlings. Proceedings of the Wildlife Damage Management Conference 11:202-208.

JOHNSON, S. R., AND I. M. COWAN. 1974. Thermal adaptation as a factor affecting colonizing success of introduced Sturnidae (Aves) in North America. Canadian Journal of Zoology 52:1559–1576.

JOHNSON, R. J., AND J. F. GLAHN. 1994. European starlings. Pages E109-E120 *in* S. E. Hygnstrom, R. M. Timm, and G. E. Larson, editors. Prevention and control of wildlife damage. University of Nebraska-Lincoln, Lincoln, Nebraska, USA.

KENNEDY, E. D., AND H. W. POWER. 1990. Experiments on indeterminate laying in house wrens and European starlings. Condor 92:861–865.

KESSEL, B. 1953. Distribution and migration of the European starling in North America. Condor 55:49–67.

KESSEL, B. 1957. A study of the breeding biology of the European starling (*Sturnus vulgaris*) in North America. American Midland Naturalist 58:257–331.

KOENIG, W. D. 2003. European starlings and their effect on native cavity-nesting birds. Conservation Biology 17:1134–1140.

LOWE, S., M. BROWNE, S. BOUDJELAS, AND M. DE POORTER. 2004. 100 of the world's worst invasive alien species: a selection from the global invasive species database. The Invasive Species Specialist Group (ISSG) a specialist group of the Species Survival Commission (SSC) of the World Conservation Union (IUCN). www.issg.org/booklet.pdf.

MATTHEWS, P. R., AND A. MCDIARMID. 1979. The production in bovine calves of a disease resembling paratuberculosis with a *Mycobacterium* sp. isolated from a woodpigeon (*Columba palumbus).* Veterinary Record 104:286.

MEIJER, T. 1990. Incubation development and clutch size in the starling. Ornis Scandinavica 21:163–168.

MORRISON, D., AND D. F. CACCAMISE. 1990. Comparison of roost use by three species of communal roostmates. Condor 92: 405–412.

NCBA. 2004, posting date. A basic look at E. coli 0157.

OTT, S. L., S. J. WELLS, AND B. A. WAGNER. 1999. Herd-level economic losses associated with Johne's disease on U.S. dairy operations. Preventive Veterinary Medicine 40:179-92.

PILCHARD, E. J. 1965. Experimental transmission of transmissible gastroenteritis virus by starlings. American Journal of Veterinary Research 114:1177-1179.

PIMENTEL, D., L. LACH, R. ZONIGA, AND D. MORRISON. 2000. Environmental and economic costs of nonindigenous species in the United States. BioScience 50:53-65.

PINXTEN, R., L. VAN ELSACKER, AND R. F. VERHEYEN. 1987. Duration and temporal pattern of mate guarding in the starling. Ardea 75:263–269.

POWER, H., E. D. KENNEDY, L. R. ROMAGNANO, M. P. LOMBARDO, A. S. HOFFENBERG, P. C. STOUFFER, T. R. MCGUIRE. 1989. The parasitism insurance hypothesis: why starlings leave space for parasitic eggs. Condor 91:753–765.

RICKLEFS, R. E., AND C. A. SMERASKI. 1983. Variation in incubation period within a population of the European starling. Auk 100:926–931.

ROMAGNANO, L., A. S. HOFFENBERG, AND H. W. POWER. 1990. Intraspecific brood parasitism in the European starling. Wilson Bulletin 102:279–291.

ROYALL, W. C., JR. 1966. Breeding of the starling in central Arizona. Condor 68:195–205.

ROYALL, W. C., JR., T. C. DECINO, AND J. F. BESSER. 1967. Reduction of a starling population at a turkey farm. Poultry Science 46:1494-1495.

STORCH, G., J. G. BURFORD, R. B. GEORGE, L. KAUFMAN, AND L. AJELLO. 1980. Acute histoplasmosis. Description of an outbreak in northern Louisiana. Chest 77:38-42.

SULLIVAN, H., G. M. LINZ, L. CLARK, AND M. SALMAN. 2006. West Nile virus antibody prevalence in red-winged blackbirds (*Agelaius phoeniceus*) from North Dakota (2003-2004). Journal of Vector-borne and Zoonotic Diseases 6:305-309.

SUTHERS, H. B. 1978. Analysis of a resident flock of starlings. Bird-Banding 49:35–46.

TINBERGEN, J. M. 1981. Foraging decisions in starlings (*Sturnus vulgaris*). Ardea 69:1–67.

U.S. DEPARTMENT OF AGRICULTURE. 2000. Environmental assessment: bird damage management at livestock feeding facilities in the Kansas Wildlife Services program. U.S. Government Printing Office, Washington D.C., USA.

U.S. GEOLOGICAL SURVEY. Patuxent Wildlife Research Center. 2005. North American breeding bird survey internet data set, 7 October 2005 (www.pwrc.usgs.gov/bbs/retrieval/).

WEBER, W. J. 1979. Health hazards from pigeons, European starlings, and English sparrows. Thompson Publications, Fresno, California, USA.

WEST, R. R. 1968. Reduction of a winter starling population by baiting its preroosting areas. Journal of Wildlife Management 32:637-640.

# CONTROL AND ERADICATION OF THE NORTH AMERICAN RUDDY DUCK IN EUROPE

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***Abstract:*** The ruddy duck (*Oxyura jamaicensis*) was introduced to wildfowl collections in the United Kingdom (UK) in the 1940s and subsequently escaped into the wild. Estimated numbers have increased rapidly in the UK from 20 wintering birds in 1962 to 5,946 in January 2000. As the population of feral ruddy ducks in the UK rose, so did the number of records of ruddy ducks on the European mainland. In 1984, ruddy ducks were first recorded in Spain, which is home to internationally important populations of the rare indigenous white-headed duck (*Oxyura leucocephala*). By 1991, hybrids resulting from crosses between the two species were recorded in that country. The two species hybridise readily, with the ruddy duck apparently possessing the competitive advantage. The ruddy duck is now recognised as the most significant threat to the white-headed duck and the UK appears to be the main source of birds reaching mainland Europe. This paper details the development of a control and then an eradication programme for this species in the UK.

***Key Words:*** invasive species, non-native, *Oxyura*, ruddy duck, United Kingdom, white-headed duck.

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## INTRODUCTION

The control of established non-native species provides a range of practical, political and logistic problems. The United Kingdom (UK) has a history of eradicating invasive non-native terrestrial vertebrates such as the coypu (*Myocastor coypus*), muskrat (*Ondatra zibethicus*), and porcupine (*Erethizon dorsatum*) to protect farming and commercial interests (Gosling and Baker 1989). This paper describes the control and ongoing eradication programme directed at the North American ruddy duck (*Oxyura jamaicensis*) in the UK. This is an unusual programme given that the target is a bird and effort is spread throughout the UK. It is also unusual that the motive is to protect international biodiversity and, in particular, the status of the white-headed duck (*Oxyura leucocephala*) in Spain. The ruddy duck programme has evolved through a number of discrete stages, each with separate political, economic and practical constraints. This paper details the development of this project, and the process by which the methods and costs have been refined to produce the current eradication programme.

## THE WHITE-HEADED DUCK

The white-headed duck is the only stiff-tail duck indigenous to Europe. In the breeding season, it is largely confined to freshwater or brackish, alkaline, eutrophic lakes. Breeding sites are small or enclosed in areas within larger wetland systems. They are typically shallow and fringed with dense emergent vegetation which holds the nest sites. Wintering sites are generally larger, deeper and often have little emergent vegetation. The principal food source is chironomid larvae but a variety of other invertebrates, seeds and plants are eaten (Green and Hughes 2001).

(a) The only European breeding population occurs in Spain, where it is strictly protected. Numbers of white-headed ducks in Spain have risen from 22 in 1977 to 2,600 in early 2003 following the introduction of measures to protect their habitat and a ban on hunting of the species. Over 82% of breeding and wintering white-headed ducks in Spain occur on protected sites.

## THE INTRODUCED RUDDY DUCK

The ruddy duck is a native of North America and Central America, and the northern part of South America. In its native range, it is common, with a stable population of over 500,000 (Wetlands International 2002). It breeds in emergent

vegetation on eutrophic water bodies of all sizes. Outside the breeding season, it tends to occur in large flocks on larger water bodies, particularly artificial reservoirs in the UK. In its native North America, very large numbers winter off the coasts of the United States and Mexico but birds are only very rarely observed under these conditions in the UK.

Over the last 50 years, ruddy ducks have established a feral population in the UK and are now beginning to colonise other north-western European countries. A small number were brought to the UK and introduced into waterfowl collections in the 1940s. However, approximately 70 individuals escaped or were released in the 1950s, allowing the establishment of a feral population. By January 2000, the UK population was estimated at 6,000 birds (Wetlands Advisory Service 2002). In 1965, the first European record of a free-flying ruddy ducks outside the UK was reported.

Outside the UK, there is only one other European country which could be considered to have a self-sustaining population of ruddy ducks at present (France), although annual breeding attempts have occurred for a number of years in Ireland and the Netherlands. Even so, numbers of breeding birds in all of these countries remain very low compared to numbers in the UK, which still holds over 95% of the total European population. Hughes et al. (2004) estimated that France has around 20 breeding pairs and the Netherlands 4-7 breeding pairs. In Ireland (excluding Northern Ireland), there have been only four confirmed breeding records since 1992. However, breeding is probably more regular than this suggests. No other European countries have regular (annual) breeding records for ruddy ducks, although many countries have occasional records.

It is still highly likely that immigration of ruddy ducks from the UK to France, the Netherlands, and Ireland is occurring. For example, peak winter counts in France recorded almost 200 birds in 2003-2004 (Hughes et al. 2004) but there are only an estimated 20 breeding pairs during the summer. The increase in the UK population has been highly correlated with the increase in records for continental Europe, even to the extent that the number of continental records was lower in years following short-term declines in the UK population (Hughes et al. 1999). It is believed that all ruddy ducks (both feral and captive) in Europe are descended from the few birds imported into the UK in the 1940s. These birds have a low genetic variability compared to wild birds in their native range. DNA analyses of ruddy ducks shot in Spain, France, and Iceland strongly suggest that these individuals do not originate from North America because of their low genetic variability. The DNA fingerprints of these birds were, however, very similar to captive and feral ducks from the UK.

Ruddy ducks have now been recorded in 23 Western Palearctic countries with breeding records in at least 11, and regular breeding attempts in five (France, Ireland, Morocco, Netherlands, and the UK).

## HYBRIDIZATION

Genetic research (McCracken et al. 2000) shows that ruddy and white-headed ducks are separate species which have been geographically isolated without any gene flow for between two and five million years. In 1982, ruddy ducks were first recorded in Spain and this species has now been recorded annually since 1991. At least 139 individuals have been recorded in a minimum of 43 different locations in Spain. Despite an active and well-organised control programme to cull ruddy ducks, 59 hybrids have been recorded on at least 23 sites (Hughes et al. 1999). The mating strategies of the two species are distinct and the male ruddy duck has an apparent mating advantage over the male white-headed duck. Male ruddy ducks are not territorial, but rather defend "revolving territories" around their mates. Unpaired males attempt persistently to mate with females of both their own and other species. Forced copulation by one or more males on lone females is common (Gray 1980). In contrast, male white-headed ducks form hierarchies, with dominant males forming stable pair bonds with one or more females (Torres et al. 1985) and defending territories of emergent vegetation (Amat and Sanchez 1982). In captivity, ruddy duck x white-headed duck hybrids are fertile to at least the third generation.

There is a degree of uncertainty about how the genetic dynamic between the two species would develop if larger numbers of ruddy ducks were allowed to reach Spain. However, given the aggressive and promiscuous mating strategy of the ruddy duck, the extent of hybridization is likely to increase.

## THE DEVELOPMENT OF RUDDY DUCK CONTROL IN THE UK

### The Feasibility Study 1993-1996

In 1993, a project was commissioned to undertake small-scale research into the feasibility of control, using a variety of methods. This project, costing £104k ($208k), was carried out by the Wildfowl and Wetlands Trust (Hughes 1996). The project found that breeding season shooting was the most effective method of control, followed by winter shooting from the bank. Egg-oiling and nest-trapping of females were also successfully carried out, but these methods proved less effective than shooting. The conclusions drawn from this work were that eradication was feasible, but that larger scale control was required to more fully define the timescale and costs involved.

### The Regional Control Trial 1999-2002

The Ruddy Duck Regional Control Trial was established in April 1999 and ended in May 2002. The full report is available on the web (www.defra.gov.uk/wildlife-countryside/scientific/ruddy/ruddy1/Fullreport.pdf). This trial set out to determine the feasibility, costs and access requirements necessary to control the UK ruddy duck population. The project employed a project manager together with six control officers, although not all of these worked full-time on the project. Control was primarily through the use of firearms, principally Remington .223 rifles and five-cartridge semi-automatic 12-guage shotguns, although traps were also used to capture birds for humane dispatch on a sample of sites. Shooting was conducted both from the shore and from boats, with small teams operating independently through most of the breeding system, but with all of the control officers working together on the larger wintering sites.

Three regions were selected to represent different challenges representative of the national situation. On Anglesey (a rural island on the north coast of Wales) and in the Western Midlands (a heavily populated region in central England), control took place year-round, while in Fife (a rural county in Scotland), it was limited to the autumn and early winter. These areas are believed to have held around 15% of the UK population (900 birds) at the time the trial began.

Permission from landowners to carry out control of ruddy ducks was sought on a voluntary basis for 153 sites. These sites had 193 owners/occupiers which had to be approached. Of the 193 owners and occupiers contacted, 58% gave permission for the control of ruddy ducks. Permission to carry out control was granted for 52% of the 153 sites. Control by shooting was allowed on 48% of all sites, with control by trapping on an additional 4%.

The UK ruddy duck population in January 2000 had been estimated at 6,000 birds, with a 95% confidence interval from 5,407 to 6,733 (Wetlands Advisory Service 2002). We culled 2,651 ruddy ducks over the three years, although the project was suspended for four months in 2001 during the Foot and Mouth Disease outbreak. The total included 751 females, 1,137 males and 763 immature birds.

On Anglesey, our aim was to reduce the breeding population by a minimum of 70% within three years. The original breeding population of 200 birds was reduced by over 70% within the first twelve months of the trial and by an estimated 93% within sixteen months. In the Western Midlands, the aim was to reduce the immediate pre-breeding population by the maximum amount possible. Counts on a sub-set of 17 sites showed reductions of 28% in the first twelve months. Counts on a sub-set of 23 sites showed a further 54% reduction in the second twelve months of the trial. These figures represent an overall reduction of 66% in the first two years of the trial. Our aim in Fife was to kill the maximum number of the postbreeding (autumn) population. We removed 216 ruddy ducks in Fife during the trial: 33 in 1999, 163 in 2000, and 20 in 2001.

Control during the breeding season was carried out by shooting on both Anglesey and in the Western Midlands. We made 249 control visits at this time of year and 847 ruddy ducks were shot (32.0% of the total number of ruddy ducks killed during the trial). Control of birds early in the breeding season, especially of adult females, is important to minimize the numbers of young birds hatched. It is more efficient early in the season as by late-May more cover is available for both sexes and the females begin to spend a large part of their time sitting on eggs and are thus more difficult to shoot. Typically, field officers worked in pairs at this time of year and most birds were shot from the bank with either a .223 rifle or a shotgun. On average 47% of the ruddy ducks present on breeding sites were killed per visit with a staff input of 2 hours on site per bird killed.

The much reduced ruddy duck numbers on Anglesey during the 2000 breeding season allowed an assessment of the likely time requirements when dealing with very low numbers of birds. The count information from Anglesey suggested that reduced numbers of birds are not distributed across all the

potential breeding sites. Rather, the birds appear to concentrate in the best breeding habitats in an area. In the case of Anglesey, we found ruddy ducks on around 12 sites in June 1999, with estimated population of adult birds of about 100. The average amount of staff time on site per bird killed (53 in total) was 1.0 hours during this period. In June 2000, counts suggested birds were present on only seven sites for most of the month, and the population varied between 14 and approximately 30. During this period, the average staff time per bird killed (14 in total) was 4.2 hours. Around half of this difference is due to the higher number of visits in 2000 when a visit was made for the purpose of control but either no birds were seen (although they were believed to be present) or no opportunity arose and no shots were fired.

Post-breeding (autumn) control by shooting was carried out in all trial areas while winter control was carried out in the Western Midlands and Anglesey. We shot 1,787 birds (67.4% of the total number culled) at this time of year on 17 sites. Four to six field officers were usually involved and the ducks were either herded towards guns on the bank by means of a boat or, on the larger waters, shot from the boats themselves. With only a few exceptions, shotguns alone were used at this time of year. On post-breeding and wintering sites 1 $km^2$ or less in extent, 54% of birds present were shot per visit on average, with a staff input on site of 1.1 hours per bird killed. On larger waters the percentage of birds killed was reduced (mean 19%), but the staff input on site was only 0.8 hours per bird.

Three traps were constructed at three post-breeding/wintering sites and fourteen traps on three breeding sites. Approximately 900 hours of staff effort in construction, maintenance and driving of ducks during the autumn and winter failed to result in any captures during this period. During the breeding season, approximately 750 hours of staff effort in construction and checking of traps resulted in a total of 17 ruddy ducks (five females and twelve males) being caught on one of the three sites. The results of this work suggest that post-breeding and winter trapping is ineffective with this species, but that breeding season trapping, although much less efficient than shooting, may be effective on certain sites.

We shot six non-target species during the regional trial. Three non-target ducks were shot in error. The remaining three incidents involved a swallow (*Riparia* spp.) and two coots (*Fulica atra*) flying into the line of fire. Additionally, a coot was killed by a dog collecting ruddy duck carcasses. These seven non-target casualties during shooting operations compare to 2,634 ruddy ducks shot, a non-target percentage of less than 0.3. During trap testing in spring 2002, seven young mallard (*Anas platyrhynchos*) chicks were killed when they became trapped in netting. A black-headed gull (*Larus ridibundus*) carcass was found floating in one trap although the circumstances of its death are unclear as the trap was open at the time and, having no roof, the bird should have been able to escape. This figure of eight non-target casualties compares to 17 ruddy ducks caught, resulted in a non-target kill of 47%.

A stochastic Monte Carlo simulation model was constructed to project the national ruddy duck population from January 2000 under a variety of control strategies. Three variables were included; efficacy per person (by how much each member of staff could reduce the national population by per year), numbers of staff, and changes in ruddy duck population growth rate. There could be as many as 1,000 breeding sites nationally, but it is access to 40 or so key post-breeding and wintering sites which will be critical to the acceptable progress of an eradication scheme. If this were available, modeling suggested that there was an 80% certainty that the population could be reduced to fewer than 175 birds in between four and six years, at a cost of between £3.6m and £5.4m ($7-11m) (Smith et al. 2005)

The final report on the control trial was published in July 2002. In March 2003, the Minister responsible for nature conservation and biodiversity announced that eradication of the North American ruddy duck from the UK was the Government's preferred outcome. In confirming this decision, the Government also concluded that further research work into control techniques was still required, that the protection provided by domestic legislation should be removed, and that action by other European countries was also required to remove the threat from the ruddy duck.

**National Control Trials 2003-2006**

Following the regional control trial, work continued to refine and improve methods, to extend the work to a national scale and to bid for funding from Europe. The research into the refinement of control methods on wintering sites, in particular using more staff and boats, led to a substantial increase in efficiency, particularly on larger wintering sites (i.e., those with a water surface greater than 1 $km^2$). Ruddy ducks are highly

visible, particularly in the autumn and winter, when very large numbers of birds congregate on known traditional sites. For example, in January 2000, 83% of the UK population was recorded on only 25 sites, with 67% occurring on only ten sites. This makes control easier and makes an estimate of the national population more accurate. The proportion of birds culled per visit on larger sites was 58% higher in 2003-2004 than during the Regional Control Trials. On smaller wintering sites, the proportion of birds culled was 18% higher. This significant increase in the efficiency of control operations resulted in a reduction in the predicted cost of an eradication programme of around 30%.

It has proved possible to cull between 20% and 60% (mean 41%) of ruddy ducks per visit on larger wintering sites, representing 50-110 birds per control visit in 2003-2004 (mean: 76 birds per control visit). On smaller wintering sites, the mean proportion culled per visit rises to 59% (up to 220 birds), while on breeding sites the mean proportion culled per visit is 47%.

Access has been allowed to a high proportion of sites, with 66% of site owners approached giving permission for control by shooting. More importantly, the owners of 37 of the 40 most important wintering sites in the UK have been approached in recent years, and 78% of these have given permission to shoot birds on their sites. This equates to access to over 80% of the UK wintering population. Analysis of count data shows that ruddy ducks move freely between sites in response to changes in weather conditions and as part of seasonal migration. This makes access to all sites unnecessary as it is highly likely that birds will occur at some point on sites where permission to carry out control has been granted. Between September and January, the range and number of sites holding birds is reduced as they move south and move from a larger number of smaller waters (typical of breeding and post-breeding sites) to a smaller number of larger ones (typical of wintering sites). The reverse pattern occurs in spring. It is also known that ruddy ducks move between sites in response to changes in winter weather conditions. During cold weather, birds move from smaller waters to larger ones, which are less likely to freeze over. Once milder conditions return, flocks of several hundred birds have been known to leave larger sites and to settle on other nearby waters. During the breeding season, movement between sites also occurs as a result of their mating strategy, when unattached males move between sites in search of females. As a consequence, access to all sites in an area is not essential to the programme.

Public reaction to control of ruddy ducks in the UK is mixed, but opposition has been much less than expected. Control has now been carried out on a wide range of sites across the UK, including areas where the subject is relatively emotive, such as the Western Midlands, an urban area where the regional bird club had the ruddy duck as its emblem species. Since 1999, only one control visit has been curtailed due to the presence of people opposed to the work. Only one visit has ever been cancelled as a precautionary measure – in 2001, when details of the time and location of a visit were passed to the local press.

Further refinement of the model has taken place to incorporate the new figures on control efficiency. This suggests that eradication from the UK is feasible as part of a five-year control programme. The mean time predicted to reduce the population to less than 50 individuals (i.e., by over 99%) is five years if eight staff are employed full-time for counts and control measures.

**UK Eradication 2005-2010**

In late 2005, funding was received from the European Union (EU) LIFE programme for the eradication of the ruddy duck from the UK. This source provided 50% of the costs required to undertake an eradication programme, matching funds being provided by the UK government. The objective of the project is to protect the European population of the globally-threatened white-headed duck from its most significant threat by eradicating the ruddy duck from the UK, and to exchange and disseminate information on control methods through contact with other European control teams and policy makers, leading to more effective control of ruddy ducks across Europe.

The specific actions and deliverables for this work are:

1. To remove the risk to the European population of the white-headed duck by carrying out control of the ruddy duck in the UK. Shooting will be the main method of control but others, including trapping and egg oiling, will be used as appropriate. Only cartridges containing lead-free shot and fibre wads will be used.
2. The collation of data on numbers of ruddy ducks being recorded on mainland Europe and in Spain in particular.
3. To monitor changes in the size and distribution of the UK ruddy duck population and to model the time and effort required for complete

eradication from the UK should this not be achieved within the timescale of the project.

4. To consult regularly with scientific advisors and animal welfare groups to ensure that disturbance to habitats and other species is minimised, that control and monitoring techniques are of the highest standard, and that best management practices are followed in terms of animal welfare.
5. To maintain contact with national control teams in other European countries and to disseminate best management practiced through exchange visits and workshops.
6. To raise public awareness of the need for control of the ruddy duck in order to prevent the effective extinction of the white-headed duck, with particular emphasis on site owners and other interested parties.

In the first 23 months of this programme a total of 3,637 birds have been removed.

## CONCLUSION

The problem caused by ruddy ducks has been recognised since the early 1990s and a programme of method development, feasibility assessment, modelling, cost reduction, and now, eradication has been developed over this period. This will provide the UK's largest attempted eradication programme and the only one for a bird. It will also be the first European eradication programme involving coordinated efforts from multiple countries. The work has developed though a series of short– to medium–term funding, with over three years continuity in funding only achieved in the current EU LIFE project. Control of the UK ruddy duck population is proceeding well within the current 5 year programme, although full eradication will require the control of other European populations and the management of the existing captive stocks.

## LITERATURE CITED


AMAT, J. A., AND SANCHEZ, A. 1982. Biologia y ecologia de la Malvasia *Oxyura leucocephala* en Andalucia. Donana Acta Vert. 9:251-320.

GOSLING, L. M., AND S. J. BAKER. 1989. The eradication of muskrats and coypus from Britain. Biological Journal of the Linnaean Society 38:39-51

GRAY, B. J. 1980 Reproduction, energetics and social structure of the ruddy duck. Ph.D. Dissertation, University of California, Davis, California, USA.

GREEN, A. J. AND B. HUGHES. 2001. Birds of the Western Palearctic update. Oxford University Press, United Kingdom 3:79-90.

HUGHES, B. 1996. The feasibility of control measures for North American russy duck *Oxyura jamaicensis* in the United Kingdom. Wildfowl and Wetlands Trust, Report to Department of the Environment, United Kingdom.

HUGHES, B., J. CRAIDO, S. DELANY, U. GALLO-ORSI, A. J. GREEN, M. GRUSSU, C. PERENNOU, AND J. A. TORRES. 1999. The status of the North American ruddy duck *Oxyura jamaicensis* in the Western Palearctic: towards an action plan for eradication 1999-2002. Council of Europe Publication T-PVS/Birds (99)9. Council of Europe Publishing, Strasbourg, France.

HUGHES, B., J. A. ROBINSON, A. J. GREEN, Z. W. D. LI, AND T. MUNDKUR. 2004. International single species action plan *Oxyura leucocephala*. WWT Report to Birdlife International.

MCCRACKEN, K. G., J. HARSHMAN, M. D. SORENSEN, AND K. P. JOHNSON. 2000. Are ruddy ducks and white-headed ducks the same species? British Birds 93:396-398.

SMITH, G., I. HENDERSON, AND P. A. ROBERTSON. 2005. A model of ruddy duck *Oxyura jamaicensis* eradication for the United Kingdom. Journal of Applied Ecology 42:546-555.

TORRES, L. A., C. RAYA, J. M. AYALA, AND J. M. MORENO. 1985. Estudio del compartmiento reproduction de la Malvasia *Oxyura leucocephala*. Oxyura 2:5-22.

WETLANDS ADVISORY SERVICE. 2002. The winter status and distribution of ruddy ducks in the UK, 1966/1967-1999/2000. Report to the Department for Environment, Food and Rural Affairs, United Kingdom.

WETLANDS INTERNATIONAL. 2002. Waterbird population estimates. 3rd Edition. Wetlands International. Global Series No 12. Wageningen, Netherland.

# PROBABILISTIC BIOENERGETIC/TOXICITY MODELING APPROACH FOR ESTIMATING TOXICANT INDUCED MORTALITY TO TARGET INVASIVE SPECIES AND NON-TARGET WILDLIFE

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***Abstract:*** Non-target species may be exposed to rodenticides via feeding on rodenticide baits or the carcasses of poisoned target species. As invasive species frequently negatively impact threatened or endangered (T and E) species, there is frequently spatial and temporal overlap of invasive species and T and E species. Risk assessments provide a means to estimate the probability of rodenticide associated adverse effects to non-target species (including T and E species). Quantification of risk provides critical information for decision-makers to weigh the benefits and risks of proposed rodenticide uses and to compare the risks of management with risks associated with no management (e.g. invasive species induced extinction of native species).

***Key Words:*** bioenergetic, invasive species, modeling, non-target species, probabilistic, starling, target, toxicity.

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## INTRODUCTION

Invasive species are non-native species whose introduction causes harm to the economy, environment and/or human health. In the United States, invasive species cause an estimated $120 billion in annual damage to agriculture and the environment. More than forty percent of threatened or endangered species are at risk because of invasive species (Pimentel et al. 2005). Lack of an effective invasive species management plan may lead to increased rates of extinction for native species. Chemical control is a highly effective means of reducing invasive species populations. However, the risks associated with chemical control of invasive species include potential contamination of the environment and/or potential mortality of non-target plant and animal species. Due to the temporal and/or spatial overlap of threatened or endangered species with targeted invasive species, it is imperative that the probability of adverse effects to non-target species (especially threatened or endangered species) be determined and minimized prior to the application of the toxicant. In response to this invasive species management need, we developed a quantitative approach to estimate the magnitude of toxicant induced effects (e.g., mortality) to wildlife populations of interest.

## MORTALITY ESTIMATION

Mortality is a function of exposure and a species' (or individual's) sensitivity to a toxicant. We developed probabilistic approaches to estimate exposure and sensitivity. After characterizing these two attributes, we can estimate the probability of mortality induced by a magnitude of exposure (Figure 1) (Johnston et al. 2005a, Johnston et al. 2005b, Homan et al. 2006).

### Exposure

Wildlife exposure to toxicants is often characterized as primary or secondary exposure. Primary exposure results from an animal consuming the toxicant formulation (bait). Secondary exposure results from an animal (typically a scavenger or predator) consuming another animal (or carcass) that contains residues of the toxicant. In either case, exposure is estimated by multiplying the quantity of food (or bait) consumed by the concentration of the toxicant in the food. The concentration of the toxicant in the food or bait can be determined by analytical

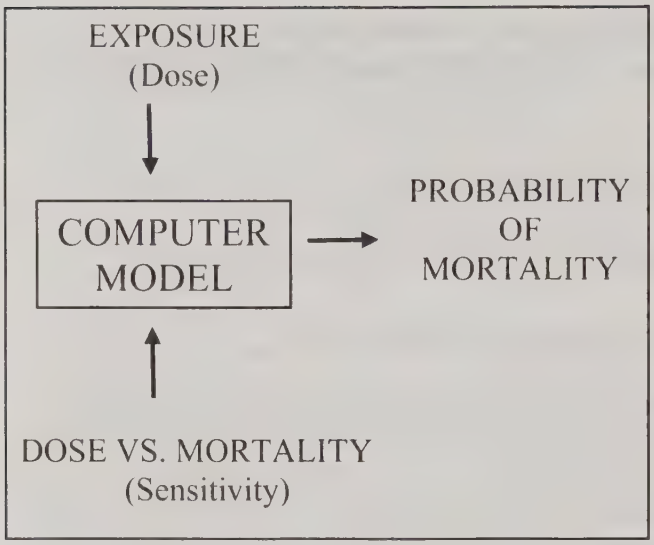



**Figure 1.** Computer modeling approach to estimate the probability of mortality based on exposure and toxicant sensitivity.

chemistry analyses. Such analyses are routinely conducted by the staff at the National Wildlife Research Center's Analytical Chemistry Laboratory in Fort Collins, Colorado.

We use a bioenergetic approach for estimating the quantity of toxicant consumed by deriving metabolic energy requirements of the target/nontarget species then dividing by the energy content of the food or bait. This approach estimates the amount of food or bait needed to satisfy the animal's daily metabolic needs, which are a function of species, activity level, body weight and environmental conditions, such as air temperature, humidity and wind velocity. The bioenergetic approach is applicable to a wide variety of species and geographic areas. It is often combined with or validated by empirical observations. Such observations may include bait consumption rates, bait feeding intervals, necropsy analyses of birds to determine the percentage or amount of a particular

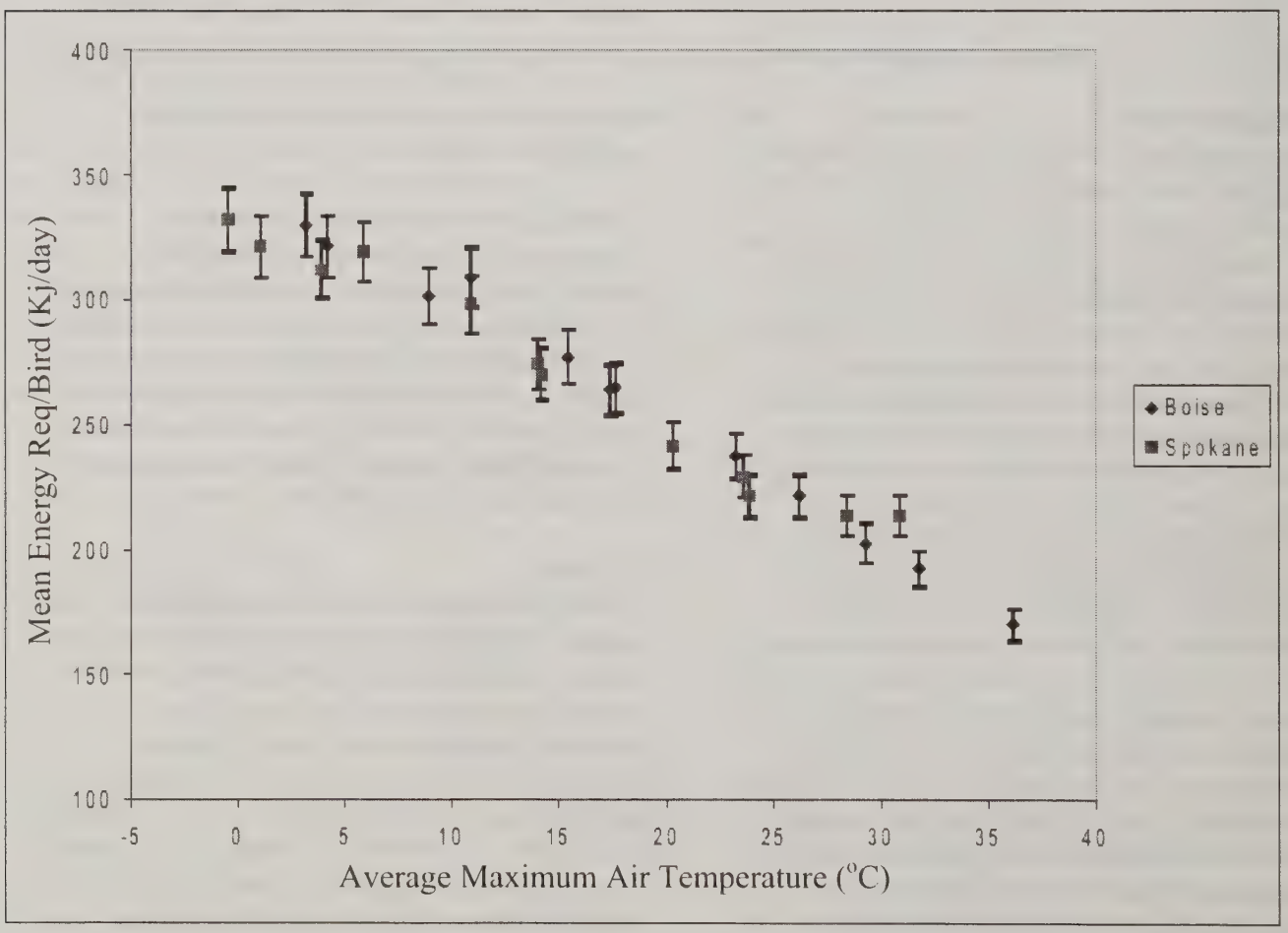



**Figure 2.** The effect of temperature on starlings' energy requirements in Boise, ID and Spokane, WA.

food source consumed and/or the amount of bait left on a plot after a feeding interval. The effect of temperature on starlings' energy requirements is summarized in Figure 2 (Stahl, unpublished data).

### Sensitivity Estimation

The sensitivity of a species or individual animal can be characterized with a dose versus mortality relationship. This relationship is typically generated with animal experiments where groups of animals are dosed with the toxicant of interest. Each group receives the same dose and the resulting percentage mortality for each group is subsequently determined. The best fitting curve is then estimated by plotting dose versus percent mortality (Figure 3).

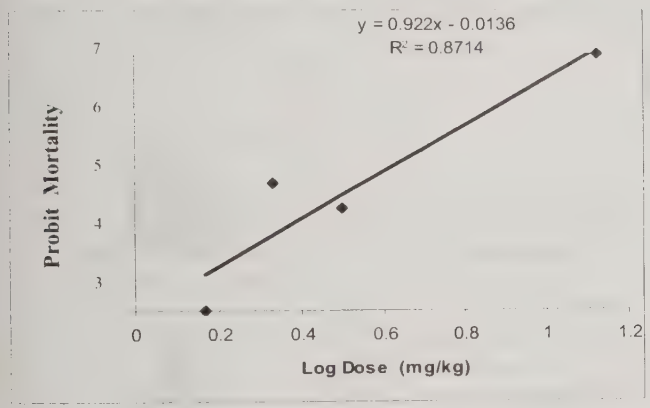



**Figure 3.** Dose versus mortality relationship is indicative of a species sensitivity to a toxicant.

### Mortality Estimation

The probability of mortality associated with any subsequent exposure can be estimated by regressing exposure (dose) against the dose versus mortality curve. The resulting estimate will be the predicted mean mortality for the dose of interest (Figure 1).

## PROBABILISTIC APPROACH

Individuals vary with respect to their sensitivity to a toxicant and their metabolic energy needs. Such variation can be attributed to a variety of factors including inter-individual variation in absorption and metabolism. The inter-individual variation (standard deviation) associated with the mean energy needs for a species can be estimated by the approach of Nagy et al. (1999). With respect to toxicant sensitivity, the result of this variation is illustrated in the magnitude of the residuals, the difference between the data points and the best fitting curve, in the dose versus mortality curve (Figure 3). The standard deviation associated with the mean LD50 and slope of the dose versus mortality curve can be estimated using the classic probit analysis approach of Finney (1971).

To capture the individual variation in exposure and sensitivity, we utilized the mean and standard deviation values to construct normal distributions of energy requirements, LD50s and dose versus mortality slopes. Probabilistic sampling (the probability of selecting individual values from these distributions are based on the frequency of each value in the distribution) for each individual in the population of interest, permitted us to construct a theoretical population which encompasses the range of exposures and toxicant sensitivities anticipated in the exposed population. Each iteration of our model represented an individual in the potentially exposed population (Figure 4). Based on the unique exposure and sensitivity assigned to each individual, the probability of mortality for each individual was estimated. The frequency of each mortality value is summarized in a reverse cumulative frequency plot (Figure 5).

## RISK MANAGEMENT

From the reverse cumulative plot, we can estimate mean mortality as well as confidence intervals associated with the estimates. These values are extremely valuable to risk managers. For example, a risk manager might use the lower 95$^{th}$ percentile mortality estimate to conservatively estimate the predicted treatment impact on the target species, while using upper 95$^{th}$ or 99$^{th}$ percentile mortality estimates to conservatively estimate the potential impact on non-target species. Linking these estimates to population models can produce a scientifically based approach for identifying chemical based invasive species management strategies with acceptable, minimal impacts on non-target species.

## LITERATURE CITED


FINNEY, D. J. 1971. Probit Analysis. 3$^{rd}$ edition. Cambridge University Press, Cambridge, United Kingdom.

HOMAN, H. J., R. S. STAHL, J. J. JOHNSTON, AND G. M. LINZ. 2005. Estimating DRC-1339 mortality using bioenergetics: a case study of European starlings. Proc. 11$^{th}$ Wildlife Damage Management Conference 11:202-208.

JOHNSTON, J. J, T. M. PRIMUS, D. J. KOHLER, M. J. HOLMES, AND A. HART. 2005a. Probabilistic model for estimating field mortality of target and non-target bird populations when simultaneously exposed to avicide bait. Pest Management Science 61:649-659.

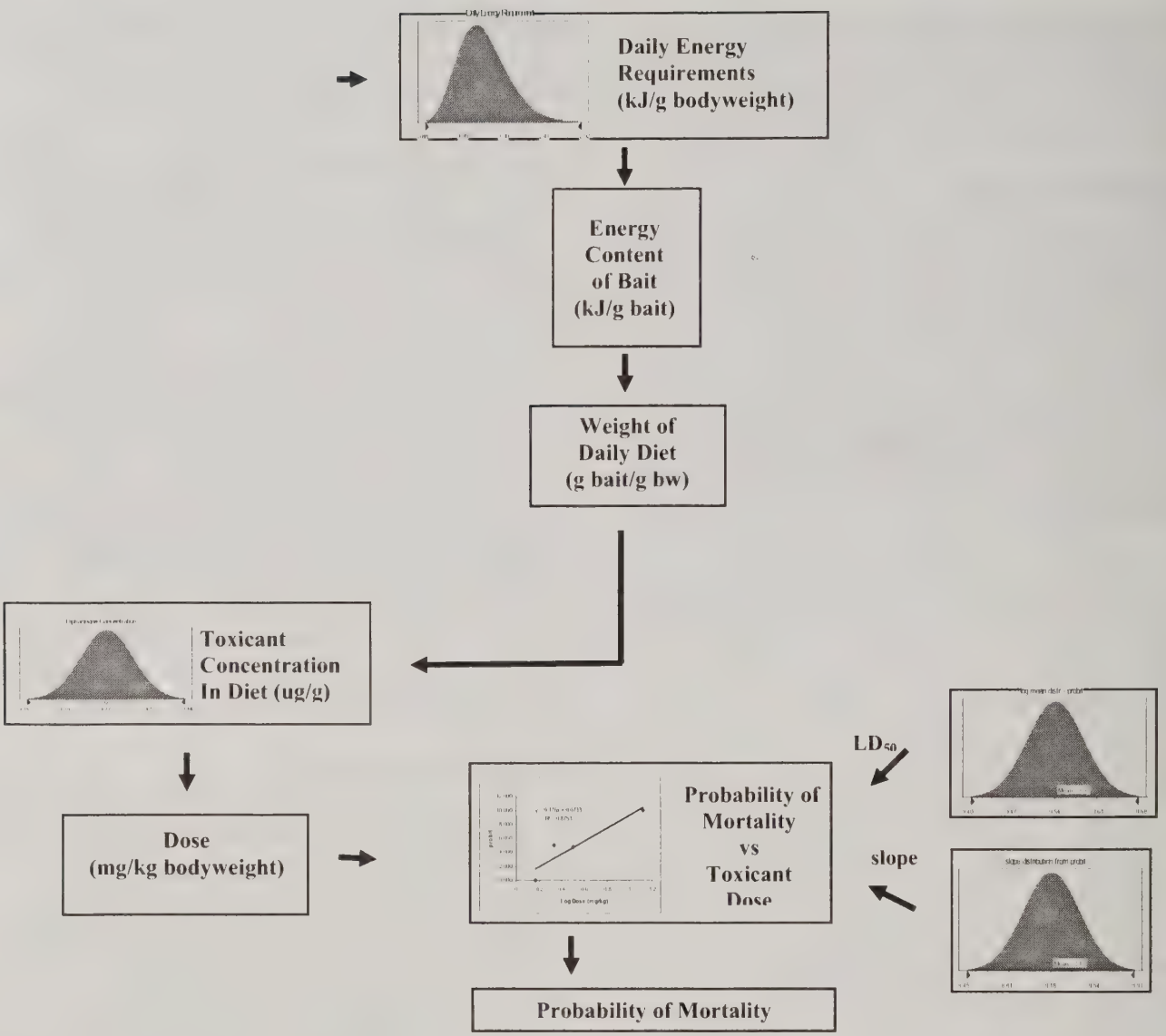
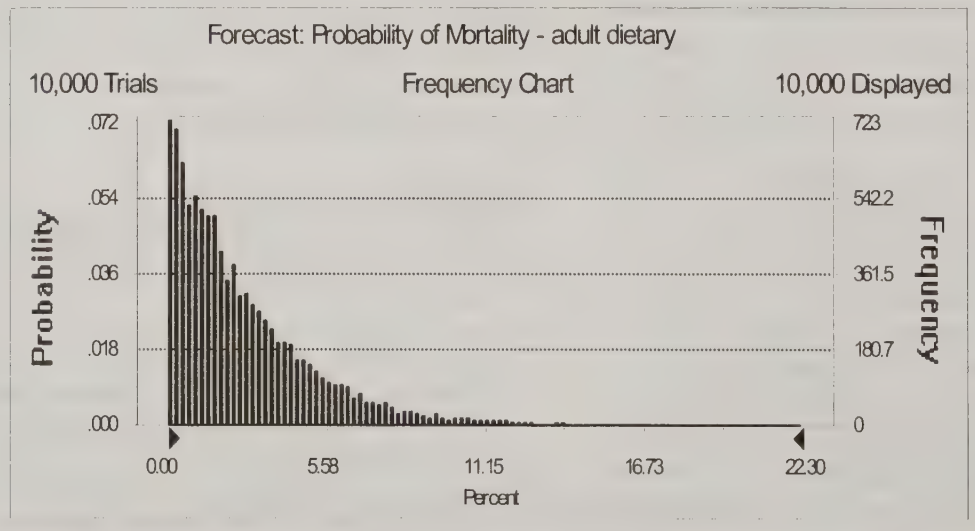



**Figure 4.** Computer model flow chart.




**Mean Mortality = 2.8%; 95% Confidence Interval = 0.24 – 11.9%**

**Figure 5.** Model output: reverse cumulative probability of mortality frequency plot.

Johnston, J. J., W. C. Pitt, R. T. Sugihara, J. D. Eisemann, T. M. Primus, J. Crocker, M. J. Holmes, and A. Hart. 2005b. Probabilistic risk assessment for snails, slugs and endangered honeycreepers in diphacinone baited areas on Hawaii, USA. Environmental Toxicology and Chemistry 24:1557-1567.

Pimentel, D, R. D., Zuniga, and Morrison. 2005. Update on the environmental and economic costs associated with alien-invasive species in the United States. Ecological Economics 52:273– 288.

Nagy K. A., I. A. Girard, and T. K. Brown. 1999. Energetics of free ranging mammals, reptiles and birds. Annual Review of Nutrition 19: 247-277.

# REDUCING RISK AND ENHANCING EFFICIENCY IN NON-NATIVE VERTEBRATE REMOVAL EFFORTS ON ISLANDS: A 25 YEAR MULTI-TAXA RETROSPECTIVE FROM SANTA CRUZ ISLAND, CALIFORNIA

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***Abstract:*** Effective conservation of native biodiversity on islands often requires the eradication of destructive non-native vertebrates. There are risks in conducting an eradication project, however, including the risk that the effort will fail to remove all the individuals, and the risk that the removal of the species will trigger ecological cascades with unanticipated and undesired consequences. Managers must plan to reduce such risks, and also maximize the return on investment of the limited conservation resources available for restoration programs. I discuss four vertebrate removal projects implemented on Santa Cruz Island, CA, over the past 25 years: sheep, golden eagles, pigs, and wild turkey. Collectively, these projects illustrate general principles for reducing risks inherent in eradication projects and for enhancing efficiencies in delivering conservation outcomes. Lessons from this case study – such as the value of disciplined engagement of the target population, strategic sequencing of restoration projects, and intensification of effort through the application of advanced technologies – can be applied to help accelerate the restoration of island ecosystems elsewhere and so the conservation of highly imperiled island biota.

***Key Words:*** eradication, golden eagle, hyperpredation, invasive species, island conservation, mesopredator release, pig, sheep, turkey.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Non-native vertebrate species can pose a significant threat to island ecosystems and it is often necessary to eradicate them to prevent extinctions of unique native biota (GISD 2007, Reaser et al. 2007). When planning and implementing an eradication effort, however, managers must contend with a variety of risks. One risk is that the removal effort will ultimately fail, due in part to the great difficulty of detecting animals at low abundance; individuals that escape detection could repopulate the island. Another risk, ironically, is a risk in success. Because eradication efforts are targeted at species posing substantive threat to island resources, their removal can cause a fundamental shift in community dynamics and perhaps ecosystem function of an island. Through such cascades, undesired effects may result. While some undesired effects might be predictable, others – owing to the limitations of our ecological understanding – may be wholly unanticipated. While island managers may be motivated by the desired effects they hope to accrue through eradication, they must also be prepared to detect and manage the undesired, and sometimes unanticipated, effects.

Santa Cruz Island, approximately 40 km off the coast of Santa Barbara, CA, USA, has been the focus of intensive restoration efforts for over 25 years. The island is comprised of two rugged mountain ranges flanking a central valley; the on-island road network is minimal and unreliable under rainy conditions. The Nature Conservancy (TNC) assumed conservation management of 90% of the island in 1978, and currently owns 76% of the 243 $km^2$ island. The United States National Park Service (NPS) now owns the remainder. NPS and TNC manage the island in partnership.

Much of the restoration effort to date has focused on removal of non-native species – legacies of an earlier ranching era. By looking across this multi-taxa and multi-decade restoration program, some strategic principles emerge for reducing risk and enhancing efficiency in vertebrate removal efforts that might have application for other island systems. This is not to say that this series of projects collectively exemplifies best practice. Rather, like all such efforts, these projects provide an opportunity to review different approaches and their outcomes, so that future

projects might adaptively benefit from that experience. The Santa Cruz Island case study does illustrate the complexity of ecological relationships that needs to be considered in order to enhance the likelihood of success in a removal effort, and to manage an island through the transition precipitated by a species' removal. This history also illuminates the value of, and approaches for, seeking enhanced efficiency in the implementation of removal programs, not only as a means to achieve better return on investment of limited conservation funds, but also as a risk reduction strategy.

## REMOVAL EFFORTS

Below, I present issues pertaining to efforts to remove four vertebrate species from Santa Cruz Island. For each, I provide brief background on the removal approach and the current status of the removal effort. In the synthesis section that follows, I discuss general observations regarding investment and ecological risk that might have application for projects elsewhere, including the role that programmatic efficiency can play in reducing such risks.

### Sheep Removal

Sheep (*Ovis aries*) were introduced to Santa Cruz Island in the 1850s, with devastating ecological consequences: destruction of unique native vegetation, destabilization of slopes, loss of soils, and more (Van Vuren 1981). The eradication of sheep on the island occurred in two phases. The first phase occurred between 1981-1989, when TNC removed over 37,000 sheep from 90% of the island, using mostly volunteer ground-based hunters working pasture by pasture (Schuyler 1993). The second phase occurred between 1997-2000 when NPS removed sheep from the remaining 10% of the island. Because those sheep were considered property of the previous owner, NPS was required to live capture them for transport to the mainland. Although it was originally estimated that approximately 3,000 sheep occurred on that portion of the island, NPS removed over 9,200. While this discrepancy may represent a difficulty of estimating populations, it might better indicate the difficulty of keeping pace with replacement using live capture as a removal technique. During the interval between the two removal phases, hundreds of sheep crossed to TNC's "sheep-free" side of the fence, and vigilant monitoring and continued hunting was required to protect the Phase I investment. In the end, it is not known how many fewer sheep might have needed to be dispatched – and how much degradation of the island could have been averted – if the program had been more intensive and accelerated.

The desired effect of the removal of sheep was recovery of native vegetation, and that recovery has been dramatic (Figure 1). But in some areas there was also a concurrent proliferation of some non-native pest plants (Klinger et al. 1994). The degree to which the extent and severity of these weed infestations can be attributed to release from sheep grazing is not known, however, because in 1988-1989 another significant modifier of habitat was removed: approximately 2,000 head of domestic cattle (*Bos taurus*) were removed via round up and transport to the mainland, and vegetation response to cattle removal can be profound (see Wagner et al. 2004). Nevertheless, the release of both desired and undesired vegetation can be categorized as an "anticipated effect" of the sheep removal project. That same vegetation recovery, however, also set in motion an effect, described below, that was unanticipated and perverse, and that will take decades to manifest.

### Golden Eagle Removal

There is no evidence that golden eagles (*Aquila chrysaetos*) were breeding residents on Santa Cruz Island prior to the 1990s. By the end of that decade, however, multiple pairs had established territories on the island. With golden eagle populations in North America increasing in recent decades, immigrants had apparently arrived on the island, where they found not only an abundant food supply of feral pigs, but also the endemic island fox (*Urocyon littoralis santacruzae*), naïve to aerial predators and generally exposed by the slow recovery from devegetation caused by earlier overgrazing of sheep and cattle. A "hyperpredation" scenario ensued, whereby the non-native pigs sustained the non-native golden eagle population, and incidental predation led to a precipitous decline of native prey, the island fox (Roemer et al. 2002; Figure 2). In 2004, the island fox was federally listed as an endangered species.

In 1999, efforts were launched to live capture and translocate golden eagles from the northern Channel Islands to the mainland (Latta et al. 2005). At that time, it was assumed that a substantial and rapid reduction in the eagle population would suffice to allow for fox viability, and that not all of the eagles needed to be removed. Since then, 32 free-flying birds have been captured, mostly using concealed, baited, and manually-triggered bownets;




**Figure 1.** Vegetation change on Santa Cruz Island, 1985-2005. Maps depict vegetation coverage, pooled into general categories: bare ground and herbaceous vegetation, white; scrub and low stature vegetation, light gray; chaparral and medium canopy communities, dark gray; forest and woodland, black. (A) Vegetation map prior to/during the eradication of feral sheep (adapted from Jones et al. 1993 and Howarth et al. 2005) (B) Vegetation map classified from a 2005 image (adapted from TNC 2007). Inset shows the island location in the state of California.

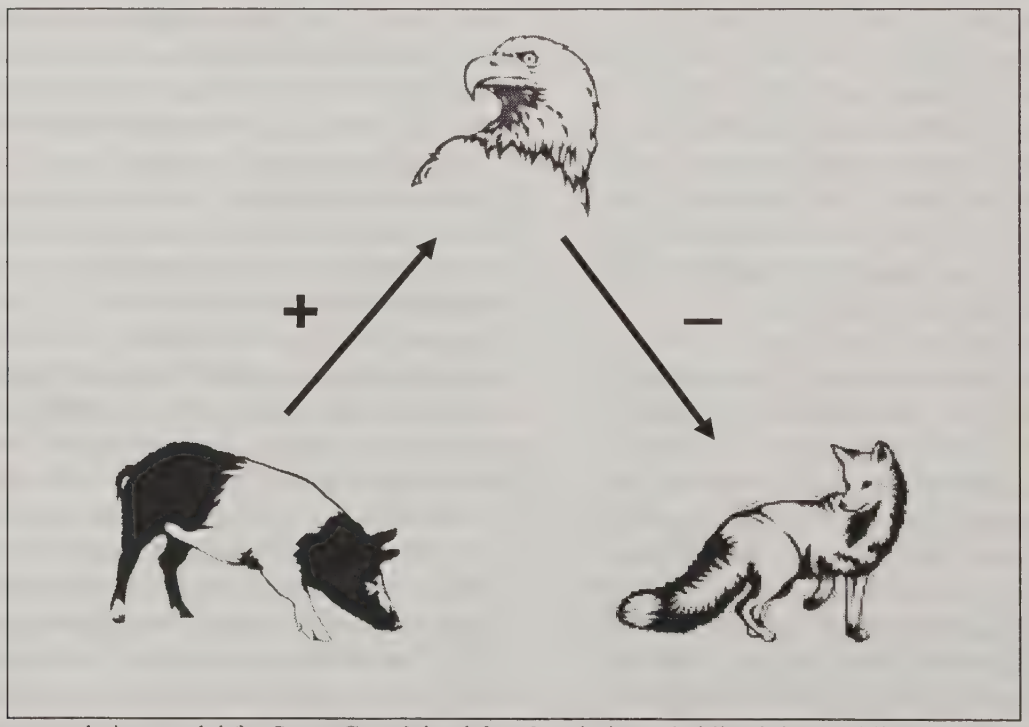



**Figure 2.** Hyperpredation model for Santa Cruz Island fox. Feral pigs subsidized the establishment of a population of golden eagles, which through incidental predation drove the endemic fox to near extinction.

eggs and young were also removed from nests (Latta 2004, IWS 2006). Attempts to capture adult eagles at nest sites using nets and other methods have generally been unsuccessful. As the golden eagle population declined on the island so did capture efficiency. Between 1999-2001, $265,000 was invested in capture efforts, netting 18 flighted birds, whereas between 2002-2004 eleven birds were captured at the expense of nearly $506,000. In 2005, with an available budget of $481,000, one eagle was captured.

It now appears that the presence of even a few eagles may slow or prevent recovery of the fox population. Between July 2005-June 2006, over 20 fox mortalities – approximately 10% of the subspecies' total estimated wild population at the time (Schmidt et al. 2007) – could be attributed to a single territorial pair of eagles. That pair was eventually captured, not by using our "traditional" methods but by employing an approach novel to the birds: a net-gun fired from a helicopter. That strategy was possible because we had a contractor on-island conducting an eradication of feral pigs, and we could couple its pilot experienced in wildlife management with a net-gun operator expert in interpreting and managing behavioral responses of helicopter-pursued eagles. That team safely captured both eagles in just two intensive half-day sessions, at a total cost of approximately $10,000.

We do not know how many golden eagles remain on the island. Sightings of eagles have become exceedingly infrequent and unpredictable. In the past year, we have had only two golden eagle observations. The mortality of radio-collared foxes has become our only indicator of continued eagle presence, and between July 2006-July 2007, we recorded 23 eagle-related fox mortalities. Ground-based and aerial surveys continue to be unsuccessful.

**Pig Removal**

Pigs (*Sus scrofa*), also introduced to the island in the 1850s, have contributed to the imperilment of nine listed plant species on Santa Cruz Island, and the endangerment of three subspecies of island fox (NPS 2002). In 2005, TNC and NPS launched an eradication effort. In contrast to the feral sheep eradication on the island, the pig eradication project was conducted by a professional vertebrate eradication contractor. The defining strategic characteristic of this effort was the eradication team's singular focus on preventing the remaining pigs from being educated to avoid hunters even as the population was steadily reduced (Morrison et al. 2007). Perhaps the most significant tactical advantage was the full integration of aerial, GIS, GPS, and telemetry technologies in the effort. A helicopter, for example, serviced almost all daily activities, which greatly enhanced efficiency on this large and rugged island (Figure 3). Nearly 80% of the 5,036 total pig dispatches were from the helicopter. With this strategic and tactical advantage the interval between the dispatch of the first and what appears to be the last pig was only 15 months.

An eradication approach of this intensity may appear cost-prohibitive to have general application. To evaluate the relative cost-effectiveness of this more intensive approach, I compared the costs of the project on Santa Cruz Island with that of an effort to eradicate pigs from neighboring Santa Catalina Island (Garcelon et al. 2005). That program began in 1990; the few pigs known to remain continue to be pursued (P. Schuyler, personal communication). A key difference between the projects on the two islands was the integral use of helicopter support on Santa Cruz. Schuyler et al. (2002) estimated the direct cost of the Catalina Island project from 1990-2001 to be $1,873,558, unadjusted for inflation and not including costs associated with fencing, fuel, and administration/support by the sponsor (Santa Catalina Island Conservancy). At the time, the authors estimated that an additional $825,000 would be needed to complete the project. From that amount I subtracted the proportion of the 1990-2001 costs that covered fuel and administration, i.e. ~11%, and added the difference to the direct costs to date, to calculate a total unadjusted direct cost of $2,604,955. For the purposes of this analysis, I assumed the Catalina Island eradication had a duration of 15 years, and divided the total direct cost by 15 to generate an average annual expenditure of $173,664 between 1990 and 2005. If each annual expenditure is adjusted to 2005 dollars (FRB 2007), the direct cost of the Catalina Island project to date has been $3,216,511. The Santa Cruz feral pig eradication "fixed price" contract value was $3,900,000, which also did not include fencing, fuel, or sponsor (TNC and NPS) administration costs. In other words, the direct cost of the Catalina Island eradication, with operations underway for approximately 15 years, is 82% the cost of the Santa Cruz Island eradication which has taken approximately two years. Interestingly, Catalina Island, at 194 $km^2$, is 80% the area of Santa Cruz Island.




**Figure 3.** Cumulative effort and outcomes from the feral pig eradication project on Santa Cruz Island, 2005-2007. (A) Black lines indicate the fence that divides the island into five pig management zones. NPS owns the easternmost zone; TNC owns the remainder of the island. Gray lines depict helicopter GPS flight paths during the hunting and monitoring phases of the project. (B) Pig dispatch locations; circles center upon trap locations with size representing the relative number of pigs dispatched in that trap. Inset shows the island location in the state of California.

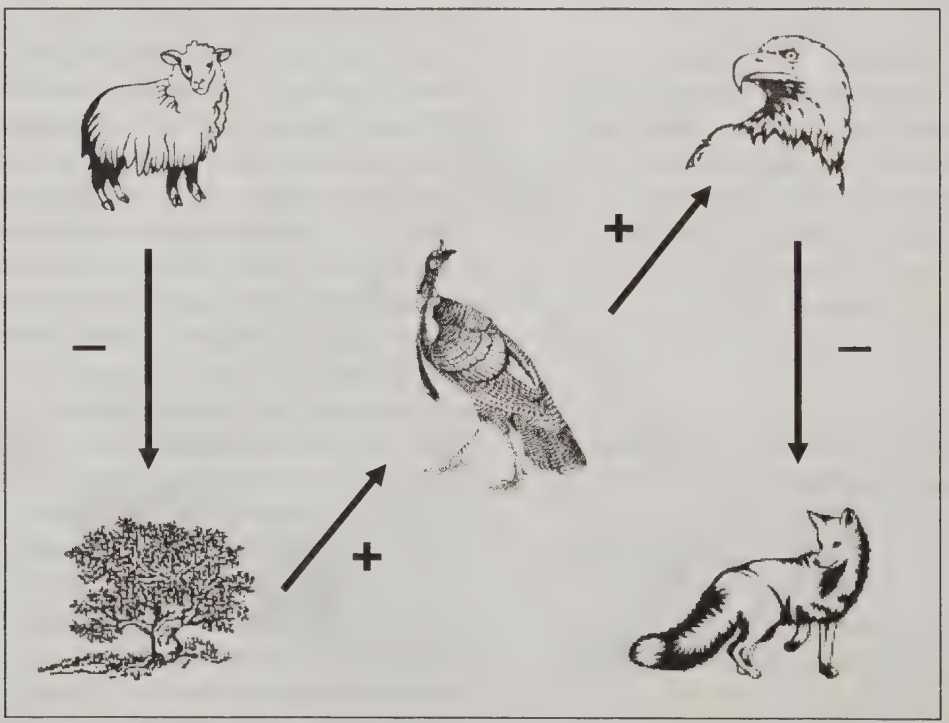



**Figure 4.** Hypothesized hyperpredation cascade for Santa Cruz Island fox. Subsequent to the removal of feral sheep, vegetation recovery improved habitat quality and quantity for introduced turkey. As the turkey population grew, it could substitute for recently-removed feral pigs as a food source for golden eagles, leading to continued imperilment for incidental prey, the island fox.

The analysis above is admittedly simplified. Costs, for example, were not distributed equally among years, and there are other differences than helicopter use between the Santa Cruz and Santa Catalina efforts (Schuyler et al. 2002). Importantly, the Catalina Island effort did not begin as an eradication project but transitioned to one after pig population control failed to produce desired and sustainable results. An earlier control phase can complicate the attainment of an eradication goal (Morrison et al. 2007). Also, unlike on Santa Cruz Island, managers on Catalina Island needed to contend with a small on-island city (Avalon, CA) that surely constrained some hunting activities. Nevertheless, the point of emphasis with the comparison is that the costs of the two projects are comparable. In the end, a less intensive approach may not necessarily be less expensive – it may just result in a different payment schedule.

While per acre direct costs may have been similar between the two islands, other associated costs were likely not. On Santa Cruz Island, *not* conducting an accelerated eradication program would have resulted in significant financial, opportunity, and ecological costs. The indirect costs (administration, operations, and other support) of maintaining an eradication team on-island would have been substantial; the sooner the project ended the sooner that capacity could be invested elsewhere. Meanwhile, the viability of many species on the island was dependent upon eradication of the pigs, so resource-intensive species-specific management efforts (e.g., island fox captive breeding, golden eagle relocation, rare plant protection) would have likely needed to be sustained as long as pigs remained. Critical restoration needs like weed control could not be addressed, not only because pigs would have likely set back any progress made, but also because species-specific crisis management left little surplus capacity to do so. With a less intensive program, investment risks also would have compounded. The longer a project takes the more vulnerable it may be to disruption by weather, waning institutional will, legal challenges, and so on. And, the longer a project takes, the more replacement and perhaps selection of the target population will occur, which can undermine the likelihood of success (Morrison et al. 2007). Simply put, the faster the eradication program, the fewer animals need to be dispatched, the lesser the myriad costs and risks, and the sooner the degradation can be halted and the restoration begun. For some imperiled species, the difference of those few years can be fateful. Thus, the question of concerning affordability of the more intensive approach should be whether the conservation community can afford *not* to use a more intensive approach.

**Turkey Removal**

In 1975, seven wild turkey (*Meleagris gallopavo*) were introduced to the island for recreational hunting. Over the following decades, the population remained localized in the center of the island and numbered approximately 40-50 (P. Schuyler, personal communication). In the early 2000s, the population began to irrupt, growing from a single overwintering flock of 46 in 1999 to a population of 276 birds in 2006, dispersed over three distinct areas (L. Laughrin, unpublished data).

What might explain the turkey population's sudden irruption? The turkey increase did correspond with a low fox population, so perhaps prior to the fox population crash, foxes provided "top-down" control of the turkey through nest and poult predation. Yet, the feral pig population was not observed to be in decline during this period, and pigs would likely depredate the nest contents of turkeys as well. A different hypothesis to explain the increase is more "bottom-up" than release from top-down control. Following the feral sheep removal, vegetation recovery on the island was extensive (Figure 1). Thus, the removal of sheep may have effectively transformed the island from poor quality turkey habitat to high quality turkey habitat. The lag between the decrease in sheep and the increase in turkeys may simply be the period of recruitment and fruiting of native vegetation, like oak (*Quercus*) species whose acorns may be an important food for turkeys.

In assessing the increase in turkey numbers, TNC evaluated the risks that turkeys posed to island biota. Of principal concern was that turkey might replace the recently eradicated pigs as a food source for golden eagles (Figure 4). Given that golden eagles do depredate turkeys (Eaton 1992), the rapid population growth and geographic expansion was alarming. Barring some rapid intervention, the turkey population would likely continue to grow, freed as it was from habitat limitation (Figure 1) and from the nest predation and food resource competition it likely faced with pigs. TNC decided that especially after such substantial investment had been made to remove pigs, it was precautionary and prudent to attempt to remove the turkeys before their population increased further. With an exponential population

growth trajectory, waiting even another breeding season could have allowed the population to increase and expand to a point that management would be significantly more difficult and expensive.

Fortuitous for TNC was that the upcoming winter – when turkeys tend to aggregate in flocks that would be amenable for trapping – was to coincide with a waning of pig hunting activity by the on-island pig eradication team. Having available much of the expertise, capacity, and equipment necessary to mobilize a turkey removal effort helped make possible the needed rapid intervention. The turkey removal was conducted with the same attributes as the pig eradication, i.e., with a focus on humane dispatch and on not educating individual turkeys as the hunt proceeded (Morrison et al. 2007). Using strategic and disciplined deployment of drop nets, the population was reduced substantially in December 2006. Some captured birds were surgically sterilized by licensed veterinarians, affixed with radio-telemetry harnesses, and released to help assess the distribution, abundance, and activity patterns of the residual population. Although it is difficult to assess progress in advance of the next overwintering season, it does appear that the program has reduced the population to a very few individuals. Cost of this program has been approximately $55,000.

## SYNTHESIS

Population removal efforts on Santa Cruz Island over the past three decades provide a unique vantage to assess risks and efficiencies in vertebrate eradication programs which may have application to efforts on other islands.

### The High Price of Education

The risk of failure in eradication programs is real and managers must explicitly focus on reducing that risk when planning and implementing their program. Not educating the remaining animals as the population is reduced is perhaps the most important means of reducing the risk of failure. That requires a focus on how the last individual will be captured – well before the first is even approached. Perhaps the greatest illustration of this principle from Santa Cruz Island comes from the golden eagle removal program. That program was initiated without an awareness that we might indeed need to capture all of the eagles. Had we that orientation early in the project, we might have been that much more cautious not to allow eagles to witness other birds being captured, or see us in the vicinity of their nest, and so on. This is not to say that the eagle teams working on the project were careless on this count. But eradication differs in a fundamental way from control: with eradication, *every* engagement with an individual matters, because ultimately every individual will need to be removed. The efficiency of the feral pig eradication is a testament to the benefit of having this strategic approach from the start. That the feral pig eradication project on Santa Catalina Island began as a population control program may to a large degree explain why it is still underway.

### Monitoring for Success

Removal of a target species may trigger ecological cascades leading to undesired or unanticipated effects on native biota. An example of an undesired but *anticipated* potential effect on Santa Cruz Island was the risk that the removal of pigs might actually increase the predation of foxes by golden eagle and so speed their extinction (Courchamp et al. 2003). We addressed this hypothesized perverse effect prior to the launch of the pig eradication by radio-collaring and monitoring a large proportion of the wild fox population, so that if the mortality rate did increase during or after the pig removal it could be detected and managed. In contrast, the turkey population increase in the wake of the vegetation recovery following sheep removal (Figure 1), and the hypothesized threat it posed for foxes (Figure 4), was for the managers on Santa Cruz Island an "unanticipated" effect. At the time of the sheep removal, golden eagles were not even considered a factor in the island's community dynamic.

Although the trophic dynamic depicted in Figure 4 may seem idiosyncratic to Santa Cruz Island, a simple substitution of species with others occupying the same trophic position (Figure 5) reveals how this potential scenario may be rather common on invaded islands of the world. What might be anticipated with the removal of a habitat modifying herbivore is an increase in food resources for smaller herbivores which may lead to an increase in their abundance and a consequent release of mesopredators from food limitation, with an adverse effect on other prey. This mechanism differs from the top-down release of mesopredators from apex predators, as described in Crooks and Soulé (1999). Indeed, most attention on food web effects of eradication on islands has focused on direct and indirect top-down effects of removal of

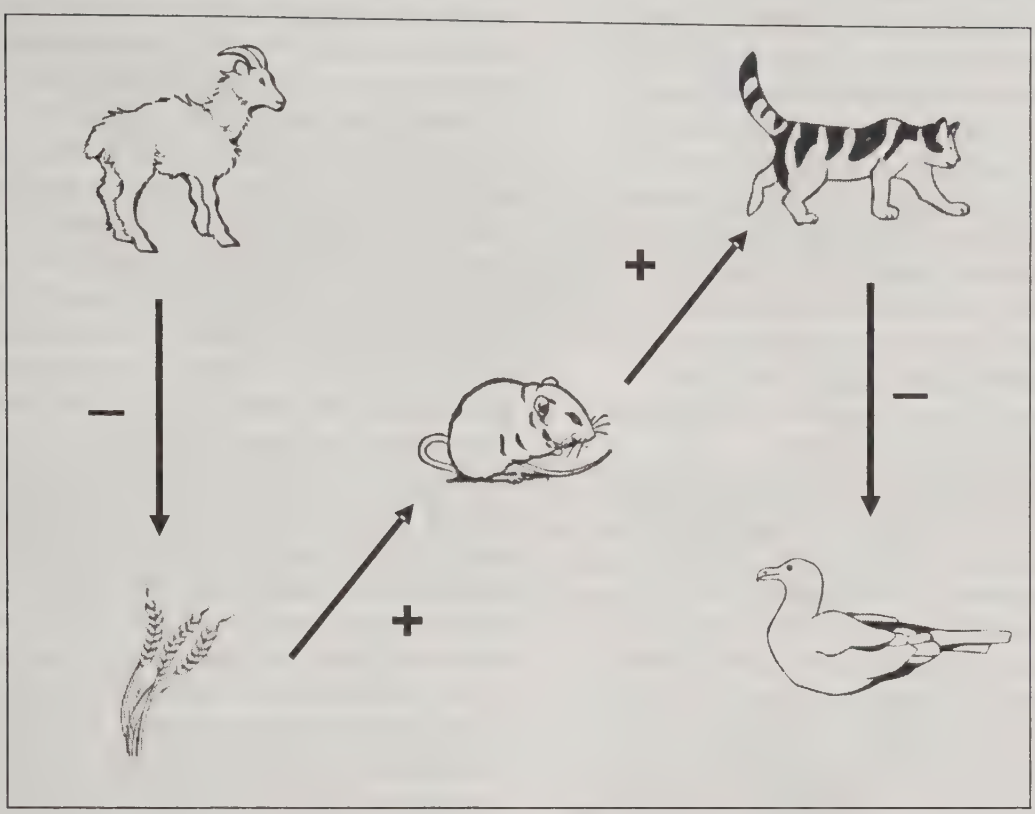



**Figure 5.** Hypothesized bottom-up release of non-native mesopredators on islands. Vegetation recovery following removal of non-native herbivores could increase the food supply (e.g., perhaps nonnative grasses) for rodents (perhaps also non-native), and that could subsidize an increase in non-native mesopredators, with a cascading adverse effect on native biota.

species (e.g., Courchamp et al. 1999, Zavaleta et al. 2001). How bottom-up processes and anthropogenic habitat change influence mesopredator release effects has not been well studied (Elmhagen and Rushton 2007). Understanding these and other ecological relationships (Courchamp and Caut 2005) is critical for managers of restoration efforts on highly degraded and invaded islands.

Managers should not be daunted by the risks of undesired and unanticipated effects, but rather be committed to lessening their number and magnitude. Above all, the examples presented here underscore the necessity of science-based planning, and the importance of ecological monitoring and responsive management – before, during, and perhaps long-after an eradication. While the dynamics depicted on Figure 5 could conceivably manifest relatively rapidly in response to herbivore eradication, it is noteworthy that the dynamics of concern with turkeys on Santa Cruz Island were taking decades to unfold.

**The Value of Trophically-Strategic Sequencing**

For islands with multiple invaders that include habitat modifying species like sheep there may be strategic advantages in eradicating other problem species first, since the removal of the grazer may trigger a vegetation recovery that could benefit the other undesired species and frustrate the removal of them. Perhaps, for example, pigs would have been easier to eradicate prior to or simultaneously with the sheep, because the devegetation wrought by sheep may have suppressed pig habitat and so pig numbers, and rendered pigs and pig sign easier to detect. Another neighboring island's feral pig eradication project may illustrate this point. Santa Rosa Island, at 215 $km^2$, was cleared of pigs between 1990-1993 using mostly ground-based hunting techniques and without the benefit of fencing (Lombardo and Faulkner 2000). At the time, Santa Rosa was largely devegetated by overgrazing of non-native herbivores; over 70% of the island was grassland or non-vegetated (Lombardo and Faulkner 2000), which is a proportion similar to the area of grassland and bare

ground on Santa Cruz Island when sheep were present, i.e., ~74% (Figure 1A). A total of 1,175 pigs were removed from Santa Rosa Island. By the time the pig eradication was underway on Santa Cruz Island, however, the proportion of open habitat (grassland and bare ground) had been very much reduced, to ~24% (Figure 1B), and a total of 5,036 pigs were ultimately removed. The direct cost of the Santa Rosa Island pig eradication (K. Faulkner, personal communication), assuming equal allocation over three years beginning in 1990 and adjusted to 2005 dollars (FRB 2007), was $1,080,050 – approximately a quarter of the direct cost of the Santa Cruz Island project.

If one species effectively limits the population size of other undesired species, managers should consider leveraging that effect to their strategic advantage. For example, in a community that has the potential to undergo a bottom-up release of predators, such as that depicted on Figure 5, there may be great benefit to first remove the invasive plant, and or the invasive rodent, and or the invasive predator before removing the species that is directly or indirectly suppressing the size of their populations and, hence, their adverse impacts. Unfortunately, trophically-strategic sequencing cannot always be implemented due to political, social or other reasons (that was the case with pigs on Santa Cruz Island; it was not our prerogative to remove them prior to sheep and cattle.) Yet, if the reasons to not optimally sequence are based more on cost and logistical considerations, those rationales should be scrutinized closely given that there may be great efficiency and economy of scale not only in optimal sequencing but also in the synchronizing of restoration efforts.

**Maximizing Restoration Return on Investment**

As was illustrated in the direct, indirect, and opportunity cost comparison between the Santa Cruz and Santa Catalina island pig eradications, projects designed for intensity and, therefore, efficiency can offer great benefit. The return on investment can be further enhanced by planning to tackle multiple issues simultaneously or in immediate succession, thus leveraging the often substantial "start-up costs" of a project. Once the pig eradication team and equipment were on Santa Cruz Island, for example, it became clear that other restoration needs could then also be met, and much more cost-effectively and efficiently than if they each required independent mobilization to the island. The golden eagle net-gunning and the turkey removal are examples of projects that had the incidental effect of maximizing the return on that initial investment in the pig eradication. By spreading costs over a variety of projects – whether by purposefully planning for and addressing multiple taxa on a single island, or the restoration needs of multiple neighboring islands – management becomes increasingly affordable. Such economies of scale can, of course, extend beyond vertebrates. Late in the feral pig eradication project, for example, TNC contracted with the provider to map weeds on the island and implement weed control on remote infestations. Use of the helicopter expedited the mapping and circumvented the risk of weed transmission by walking; it also provided additional island-wide surveillance for residual pigs (Figure 6). So a product from the weed mapping project was "free" additional monitoring for pigs. Regrettably, it was not until late in the pig eradication project that TNC began to contemplate such opportunities for efficiencies and economies of scale. It would behoove managers of future projects to proactively identify ways to leverage their investment in one aspect of restoration to benefit others.

**Efficiency as a Risk Reduction Strategy**

The longer a project takes the more it is exposed to factors that can undermine its success. The eight year lag between Phases I and II of the sheep eradication on Santa Cruz Island put the investment and accomplishment in Phase I in continuing jeopardy. The risk that sheep would reinvade the area already cleared increased with each passing year, as the contrast between vegetation conditions across the fenceline – and so the incentive to cross – increased. The feral pig eradication on Santa Cruz Island has been subject to ongoing legal challenge aimed at stopping the project. Although the plaintiffs in this case have thus far not been successful in their efforts, the potential that a program would be halted prior to its completion is real. Accelerated implementation reduces investment risk in eradication. But perhaps most importantly, efficiency can help reduce the risk of extinction of native species on islands. The demonstrated efficiency of the feral pig eradication project could be a model to help increase the pace and scale of effective biodiversity conservation on the world's islands.

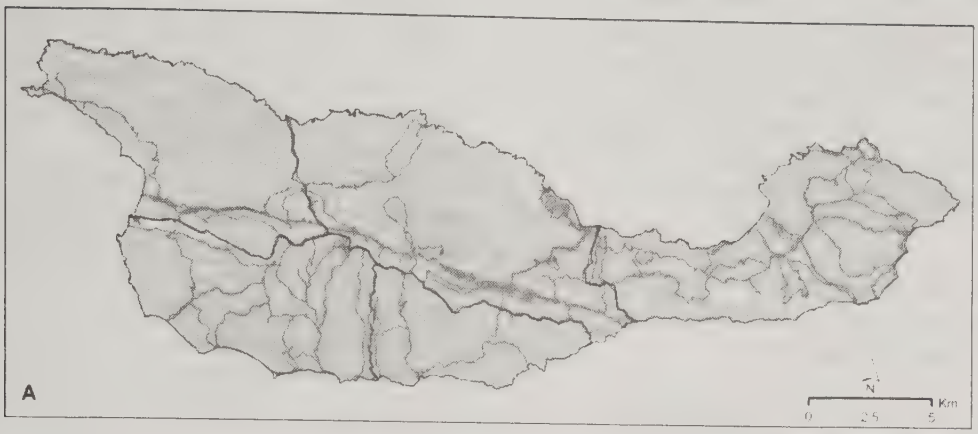



**Figure 6.** GPS tracks of helicopter surveys of 2007 weed mapping project. At the end of the feral pig eradication project on Santa Cruz Island, the contractor that had conducted the feral pig eradication won a competitive bid for an island-wide weed mapping project. Much of that work was conducted using low altitude helicopter-based surveying. The weed surveys also provided supplemental monitoring for residual pigs. The gap in coverage on the isthmus represents the no-fly zone around an island fox captive breeding facility.

## CONCLUSION

Ameliorating the extreme imperilment of island biodiversity often demands the removal of destructive non-native species. Given the scarcity of conservation resources, it is imperative that those efforts be conducted in ways that reduce their inherent investment and ecological risks, and that maximize the restoration return on those investments.

Santa Cruz Island is relatively large among the islands that have been the focus of eradication efforts (Campbell and Donlan 2005, Morrison et al. 2007), and the year 2007 marks the first in over 150 that there are no unmanaged non-native mainland vertebrates on the island. It took a quarter of a century to achieve this milestone. Over that same time period, great advances in the science and practice of eradication – theoretical, statistical, and technological advances, as well as practical experience from efforts worldwide – make it possible for work that once took decades to now take but years. These advances, and those surely to come, enable managers to now set their sites beyond incremental gains to the entirety of their island restoration goals, and implement a comprehensive restoration strategy that maximizes return on investment and minimizes the myriad risks in that investment. Restoration goals for the world's islands should, accordingly, be all the more ambitious.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

CAMPBELL, K., AND C. J. DONLAN. 2005. Feral goat eradications on islands. Conservation Biology 19:1362-1374.

COURCHAMP, F. AND S. CAUT. 2005. Use of biological invasions and their control to study the dynamics of interacting populations. Pages 253-279 *in* M. W. Cadotte, S. M. McMahon, and T. Fukami, editors. Conceptual ecology and invasions biology. Springer.

COURCHAMP, F., M. LANGLAIS, AND G. SUGIHARA. 1999. Cats protecting birds: modeling the mesopredator release effect. Journal of Animal Ecology 68:282-292.

COURCHAMP, F., R. WOODROFFE, AND G. W. ROEMER. 2003. Removing protected populations to save endangered species. Science 302:1532.

CROOKS, K. R., AND M. E. SOULÉ. 1999. Mesopredator release and avifaunal extinctions in fragmented systems. Nature 400:563-566.

EATON, S. 1992. Wild Turkey (*Meleagris gallopavo*). Pages 1-28 *in* A. Poole, P. Stettenheim, and F. Gill, editors. The birds of North America, Vol. 22. Philadelphia, PA: The Academy of Natural Sciences; Washington, DC: The American Ornithologists' Union.

ELMHAGEN, B. AND S. P. RUSHTON. 2007. Trophic control of mesopredators in terrestrial ecosystems: top down or bottom up? Ecology Letters 10:197-206.

FRB (FEDERAL RESERVE BANK). 2007. Consumer Price Index calculator. http://woodrow.mpls.frb.fed.us/research/data/us/calc/ . Accessed August 2, 2007.

GARCELON, D. K., K. P. RYAN, AND P. T. SCHUYLER. 2005. Application of techniques for feral pig eradication on Santa Catalina Island, California. Pages 331-340 *in* D. K. Garcelon and C. A. Schwemm, editors. Proceedings of the sixth California islands symposium. Institute for Wildlife Studies, Arcata, California, USA.

GISD (GLOBAL INVASIVE SPECIES DATABASE). 2007. http://www.issg.org/database. Accessed August 2, 2007.

HOWARTH, J., S. O'CONNOR AND L. HARRISON. 2005. Vegetation of Santa Cruz Island digitized from 1985 map by Steve Junak, Santa Barbara Botanical Garden. Class project from Geography 176c: GIS Design and Application. Spring 2005. University of California, Santa Barbara. California, USA.

IWS (INSTITUTE FOR WILDLIFE STUDIES). 2006. Population status and golden eagle removal efforts on Santa Cruz and Santa Rosa islands, 2005-2006. Unpublished report. Institute for Wildlife Studies, Arcata, California, USA.

JONES, J. A., S. A. JUNAK, AND R. J. PAUL. 1993. Progress in mapping vegetation on Santa Cruz Island and a preliminary analysis of relationships with environmental factors. Pages 97-104 *in* F. Hochberg, editor. Third California islands symposium: recent advances in research on the California islands. Santa Barbara Museum of Natural History, Santa Barbara, California, USA.

KLINGER, R. C., P. T. SCHUYLER, AND J. D. STERNER. 1994. Vegetation response to the removal of feral sheep from Santa Cruz Island. Pages 341–350 *in* W. L. Halvorson and G. J. Mender, editors. The fourth California islands symposium, update on the status of resources. Santa Barbara Museum of Natural History, Santa Barbara, California, USA.

LATTA, B. C. 2004. Channel Islands golden eagle translocation program, summary report, 1999-2004. Unpublished report. Santa Cruz Predatory Bird Research Group, University of California. Santa Cruz, California, USA.

LATTA, B. C., D. E. DRISCOLL, J. L. LINTHICUM, R. E. JACKMAN, AND G. DONEY. 2005. Capture and translocation of golden eagles from the California Channel Islands to mitigate depredation of endemic island foxes. Pages 341-350 *in* D. K. Garcelon and C. A. Schwemm, editors. Proceedings of the sixth California islands symposium. Institute for Wildlife Studies, Arcata, California, USA.

LOMBARDO, C. A., AND K. R. FAULKNER. 2000. Eradication of feral pigs (*Sus scrofa*) from Santa Rosa Islands, Channel Islands National Park, California. Pages 300-306 *in* D. R. Brown, K. L. Mitchell, and H. W. Chaney, editors. Fifth California islands symposium. U.S. Department of the Interior, Mineral Management Services Pacific OCS Region, Washington, D.C., Ventura, CA., USA.

MORRISON, S. A., N. MACDONALD, K. WALKER, L. LOZIER, AND M. R. SHAW. 2007. Facing the dilemma at eradication's end: uncertainty of absence and the Lazarus effect. Frontiers in Ecology and the Environment 5:271-276.

NPS (NATIONAL PARK SERVICE). 2002. Santa Cruz Island primary restoration plan. Final environmental impact statement. United States Department of the Interior. Channel Islands National Park, Ventura, California, USA.

REASER, J. K., L. A. MEYERSON, Q. CRONK, M. DE POORTER, L. G. ELDREGE, E. GREEN, M. KAIRO, P. LATASI, R. N. MACK, J. MAUREMOOTOO, D. O'DOWD, W. ORAPA, S. SASTROUTOMO, A. SAUNDERS, C. SHINE, S. THRAINSSON, AND L. VAIUTU. 2007. Ecological and socioeconomic impacts of invasive alien species in island ecosystems. Environmental Conservation 34:98-111.

ROEMER, G. W., C. J. DONLAN, AND F. COURCHAMP. 2002. Golden eagles, feral pigs, and insular carnivores: how exotic species turn native predators into prey. Proceedings of the National Academy of Sciences 99:791-796.

SCHMIDT, G. A., J. FOX, AND D. K. GARCELON. 2007. Island fox recovery on Santa Cruz Island, California, December 2000-January 2007. Unpublished report. Institute for Wildlife Studies, Arcata, California, USA.

SCHUYLER, P. 1993. Control of feral sheep (*Ovis aries*) on Santa Cruz Island, California. Pages 443-452 *in* F. Hochberg, editor. Third California islands symposium: recent advances in research on the California islands. Santa Barbara Museum of Natural History, Santa Barbara, California, USA.

SCHUYLER, P. T., D. K. GARCELON, AND S. ESCOVER. 2002. Eradication of feral pigs (*Sus scrofa*) on Santa Catalina Island, California, USA. Pages 274-286 *in* C. R. Veitch and M. N. Clout, editors. Turning the tide: the eradication of invasive species. IUCN SSC Invasive Species Specialist Group. IUCN, Gland, Switzerland and Cambridge, United Kingdom.

TNC (The Nature Conservancy). 2007. Santa Cruz Island vegetation map. http://www.tnccalifornia.org/conservation/ . Accessed August 4, 2007.

Van Vuren, D. 1981. The feral sheep of Santa Cruz Island: status, impacts, and management recommendations. The Nature Conservancy, Santa Barbara, California, USA.

Wagner, J., M. Martin, K. R. Faulkner, S. Chaney, K. Noon, M. Denn, and J. Reiner. 2004. Riparian system recovery after removal of livestock from Santa Rosa Island, Channel Islands National Park, California. U.S. Department of Interior. Technical Report NPS/NRWRD/NRTR-2004/324.

Zavaleta, E. S., R. J. Hobbs, and H. A. Mooney. 2001. Viewing invasive species removal in a whole-ecosystem context. Trends in Ecology and Evolution 16:454-459.

# DEVELOPING STRATEGY AND TOOLS FOR THE LOCAL ELIMINATION OF MULTIPLE PEST SPECIES

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***Abstract:*** Control of invasive vertebrate pests is likely to be needed in perpetuity unless their pest status changes or they are completely eradicated, both of which seem unlikely at present. This emphasises the need for pest managers to adopt long-term strategies that are both ecologically sound and cost-effective. We suggest that a strategy for simultaneous management of multiple sympatric species of pests is preferable to a single-species approach. While present strategy involves periodic control over entire areas to achieve management aims, modelling suggests that a strategy of localised elimination followed by perimeter control offers significant cost-savings in the long term. We are therefore researching three aspects of this strategy: (1) the further refinement of aerial baiting by identifying principal causes of individual pest survival, (2 the optimal deployment of control devices around the perimeter following localised elimination, and (3) the development of an efficient pest detection device to enable targeted elimination of survivors.

***Key Words:*** 1080, aerial baiting, invasive species, pest control strategy, pest detection, possum, ship rat.

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## INTRODUCTION

In New Zealand, four introduced, invasive, small mammals (the brushtail possum, *Trichosurus vulpecula*; the ship rat, *Rattus rattus*; the house mouse, *Mus musculus*; and the stoat, *Mustela erminea*) are now virtually ubiquitous. All four are seen as conservation pests because of their adverse impacts on native flora and fauna (Morgan and Hickling 2000, Innes 2005, King and Murphy 2005, Wilson et al. 2006) and possums have also become a major wildlife reservoir of bovine tuberculosis (Tb, Coleman and Caley 2000). Although these pest species have tended to be managed individually in the past, there is an increasing focus in New Zealand (and elsewhere) on managing whole ecosystems (Innes and Barker 1999), which requires integrated management of all important threats simultaneously. This automatically favours a multi-species approach to pest control (Morgan 1993) both to save money and to minimise unwanted ecological side-effects of single-species control, such as the potential for a major increase in rat numbers and impact after possum control (Sweetapple et al. 2006) or increased predation of native birds by stoats when single-species control of ship rats removed their primary prey (Murphy and Bradfield 1992).

Although these species can all be eradicated from islands or securely fenced areas, it is not currently technically feasible and economically affordable to eradicate them from large unfenced areas of the New Zealand mainland. That is because eliminating the last few pests is usually prohibitively expensive and that large expense has been considered pointless when there is a high certainty of reinvasion from adjacent unmanaged areas. Pest managers have therefore been forced to adopt a strategy of sustained control in perpetuity, with pest densities kept below some low (but not zero) threshold, below which their impacts are considered tolerable. If, however, pest numbers can affordably be reduced to zero, and if immigration can be prevented, a new strategic possibility of local elimination emerges (Morgan et al. 2006). This new strategy is, in essence, sustained control at zero density, and has three requirements: (1) an ability to cheaply reduce initially high pest densities to near zero (initial knockdown), (2) an ability to cheaply prevent all or most reinvasion (perimeter control), and (3) an ability to cost-effectively locate and eliminate the few survivors and immigrants (mop up).

In New Zealand, broad-scale pest control often involves aerial poisoning of densely forested montane areas, and is most well developed for possum control, although increasingly the same tools are being used for other pests.

**Table 1.** Aerial and ground control operations monitored since 2000 in which very low or nil survival was recorded using the residual trap catch index (RTCI, the percentage of trap-nights in which a possum was caught, NPCA 2004). Pre-control trap catch indices are typically in the range 10–60%. (Morgan et al. 2006.)

| Operation | Area (ha) | Trap-nights | RTCI % |
|---|---|---|---|
| ***a) Aerial operations*** | | | |
| Bideford – July 2000 | 2,907 | 660 | 0.0 |
| Tongariro – September 2001 | 19,980 | 1,020 | 0.1 |
| Kahutara – March 2003 | 910 | 450 | 0.0 |
| Titiraupenga – July 2003 | 10,150 | 600 | 0.0 |
| Waikaremoana – July 2004 | 9,219 | 1,170 | 0.1 |
| Kahutara – September 2004 | 1,337 | 630 | 0.1 |
| Hauhangaroa – September 2005 | 82,876 | 15,358 | <0.1 |
| ***b) Ground operations*** | | | |
| Hopkins – July 2003 | 1,500 | 2,700 | 0.0 |
| Matea – July 2003 | 14,787 | 2,550 | 0.2 |
| North Taupo – July 2003 | 2,164 | 1,740 | 0.2 |
| Hochstetter – September 2003 | 450 | 420 | 0.0 |
| Bideford – November 2003 | 1,188 | 690 | 0.0 |
| Te Wharau – April 2004 | 1,224 | 600 | 0.0 |
| Kahutara – 2004 | 2,988 | 2,250 | 0.0 |
| Holdsworth – 2004 | 1,185 | 1,800 | 0.0 |
| Raetea – 2004 | 920 | 600 | 0.0 |
| Matea – 2004 | 2,799 | 1,800 | 0.2 |

For possums, monitoring of recent control operations indicates that the first requirement of local elimination can sometimes now be met (Table 1).These outcomes prompted us first to explore whether the benefits of these highly successful knockdowns could be extended through perimeter control (Morgan et al. 2006).

We modelled the long-run cost of possum control over 60 years under four scenarios: sustained area-wide repeated control achieving either (1) a 95% reduction or (2) total knockdown (i.e. 100% reduction); sustained area-wide repeated total knockdown complemented by perimeter control reducing immigration by (3) 50% or (4) 80%.

We arbitrarily set a threshold of 1 possum/ha, which equates to an RTCI of about 5-10%, as a trigger for repeating area-wide control. Where the cost of total knockdown is similar to that of 95% control, as can be the case with aerial poisoning, the total-knockdown scenario will obviously be more cost-effective. More importantly, the model predicted that total knockdown combined with perimeter control was a more cost-effective control strategy under most scenarios than a conventional 95% control strategy even when the costs of total knockdown were up to twice as high as for the 95% control. Adding to the predicted cost savings, average possum densities would be lower under the total-knockdown scenarios, so the conservation and Tb control benefits would likely be greater.

This paper reports progress in research that has since been initiated to increase the feasibility of local elimination as a strategy not only for possums but for ships rats, house mice and stoats as well. In line with the three requirements above, we aim to:

- Increase the consistency with which pests can be knocked down to near-zero densities in a single aerial poisoning operation. Aerial control techniques have been greatly improved over the last 40 years (Morgan 2004), but while some operations are highly successful, failures still occur, so our focus is to better determine why some pests still survive.

- Determine the most cost-effective tools and tactics needed to reliably and substantially reduce immigration rates. The aim is to design low-cost perimeter control systems through bioeconomic modelling, using field data on rates of immigration and encounter with control devices when different device spacings are used.
- Development and field testing of highly sensitive but ultra-low-cost multi-species monitoring devices and systems for determining where survivors and invaders are, so that highly targeted mop-up control can be used to affordably eliminate them.

## REFINING AERIAL POISONING AS A MULTI-SPECIES TOTAL-KNOCKDOWN TOOL

Pests may survive aerial poisoning because they do not encounter any toxic bait at all, or because they encounter toxic bait but it does not contain a lethal dose, or because they encounter toxic bait but choose not to eat a lethal dosage. Research and operational
experience have led to the development of sophisticated and systematic GPS-guided bait delivery that largely eliminates the first of these risks (Morgan 2004). Pre-feeding with non-toxic bait a few days or weeks before the sowing of toxic bait has been shown to increase possum kills (Coleman et al. 2007), and sowing rate and sowing pattern obviously affect the rate at which pests encounter bait.

In 2006, we conducted a major field experiment to determine the relative effects of sowing rate (1, 2 or 5 kg of 1080-laden [sodium monofluoroacetate] diced carrot bait per hectare), sowing pattern (single direction versus cross-hatched sowing), and pre-feeding (nil, once, or twice) on the survival of possums, ships rats, and house mice. The set of 18 unique combinations of the various treatments was replicated twice in different locations a month apart, with each treatment applied to 100 ha of forest. Changes in pest abundance were monitored using changes in the rates at which pests interfered with multi-species monitoring devices, the ChewTrack Card (CTCs, described in detail below). We used 100 devices per 100-ha block to monitor interference over approximately two weeks just before and again immediately after poisoning. For the first replicate, there was no consistent effect of sowing pattern on survival, but where all of the toxic bait was sown in a single direction, a generalised linear model showed significant variation in kill for both possums ($P = 0.03$) and rats ($P = 0.046$). Pre-feeding significantly increased the kill of both possums ($P = 0.01$) and rats ($P = 0.003$) (Figure 1) and two pre-feeds appeared to be consistently more effective than one (Figure 1). Increased sowing rate resulted in a smaller, but still marginally significant ($P = 0.06$) increase in possum kill, but there was no significant effect on rats ($P = 0.36$). The second replicate appeared to show broadly similar effects of pre-feeding and sowing rate on kill, but joint statistical analysis was precluded by technical problems in the monitoring of that replicate.

The results from the first replicate, coupled with similar findings elsewhere (Coleman et al. 2007), indicate that multiple exposure to non-toxic bait is usually needed to overcome the reason(s) for possum and rat survival. In a subsequent survey of two of the blocks in early 2007, we found that surviving possums and rats still readily consumed 54% of 200 non-toxic cereal baits, but only 8% of carrot baits presented simultaneously. This near-total aversion to carrot bait implies that most, if not all, possums and rats had encountered bait during the earlier experiment and consumed some of it. Concurrent trials with penned possums indicated that only a single previous exposure to non-toxic bait any time in the preceding two months was sufficient to overcome cautious feeding effects (G. Nugent unpublished data), so we infer that the improvement resulting from a second pre-feed is, instead, a result of some change in encounter rate. Taken together, these data suggest most of the possums that survived the pre-feed treatments in the field trial did so because their first encounter with toxic bait was with a sub-lethal quantity, and that they consumed all or most of this bait, but did not find another bait before the onset of toxicosis and loss of appetite that typically occurs within 30–60 minutes of 1080 ingestion. Although efforts are made to ensure that all baits sown contain a lethal dose of toxin, fragmentation of baits can occur during their passage through sowing machinery or on impact with the forest canopy and ground, and bait size can be reduced by partial consumption by other individual pests.

## DESIGNING A COST-EFFECTIVE PERIMETER CONTROL SYSTEM

We envisage that reducing pest invasion will involve placement of pest control devices such as long-life toxic baits or kill traps at some optimal

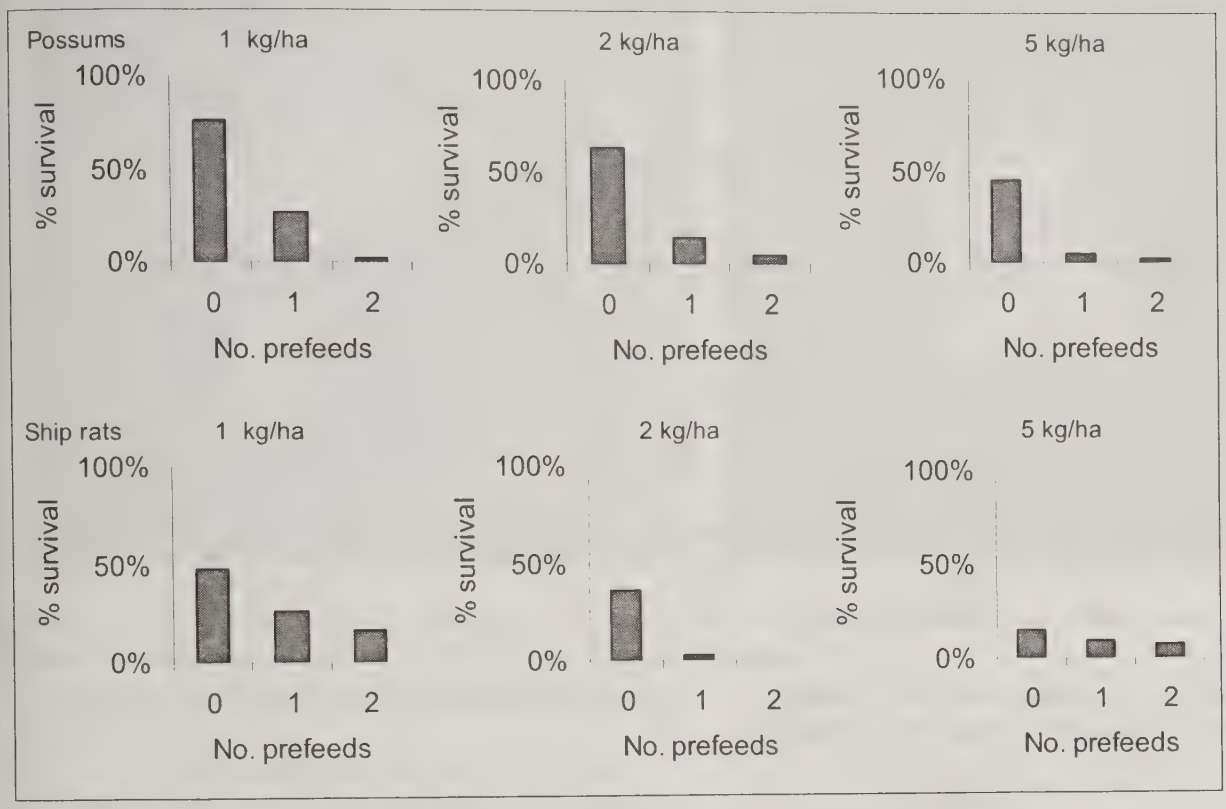



**Figure 1.** Relative measure of survival of possum and ship rat populations after aerial application of 1080-laden (0.15%) carrot baits at different sowing rates per hectare and with different numbers of non-toxic pre-feeds. The bait type sowing rate for pre-feeding treatments was the same as for the toxic treatments. The index of relative survival was estimated from changes in the Poisson-transformed detection rates partially adjusted for pre- and post-control differences in detectability.

spacing along parallel transects located near the perimeter of the management area. Identifying the most cost-efficient system for reducing invasion rates requires not only information on the cost of each device and the cost of deploying and maintaining them, but also understanding the complex interplay between device-spacing along transects, the number of transects, the spacing between transects, and their combined effect on the proportion on invader animals killed.

We have simulated these interacting influences on the proportion on invader animals killed (PIK), assuming that, even when devices are placed very close together, some dispersing pests will simply walk straight through a transect because they happened not to be hungry at the time. We also assumed that PIK would decline as the spacing was increased, and used a Weibull curve to provide an intuitively reasonable representation of that effect for a single transect, and calculated a cumulative joint probability for multiple transects (Figure 2a). The costs of perimeter control were then estimated using assumed values for the various activities involved in achieving specified levels of control (e.g., bait materials, time spent deploying baits, time spent walking between bait plots and lines, etc., Figure 2b). The particular scenario modelled in Figure 2 suggests that for a device capable of killing only 60% of invader animals when deployed at saturation levels (i.e., less than a few metres apart) approximately 80% of invader animals could be killed at the same cost using 2, 3, 4 or 5 transects. Killing an even higher percentage is predicted to require at lest three transects. Field research is now underway to paramaterise these models for possums and rats, with the major aims being to determine the "saturation" PIK for a single transect, and the shape of the curve describing how PIK declines with increasing distance between

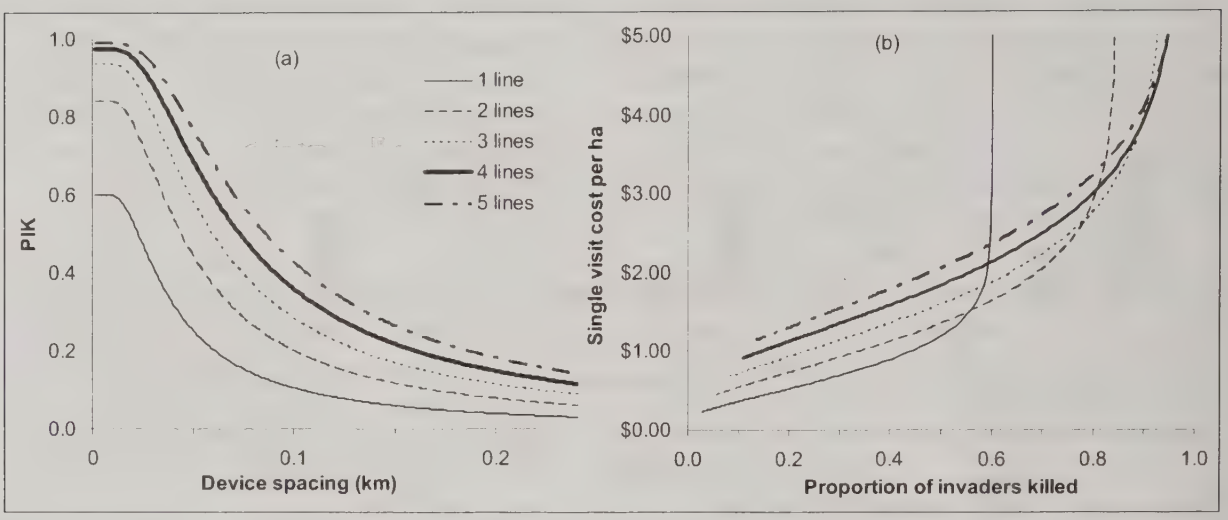



**Figure 2.** (a) Theoretical proportion of pests killed by control devices arranged at different spacings along 1-5 lines, and (b) the estimated associated costs.

devices. Using bioeconomic models, we will then be able to design cost-effective long-term strategies for perimeter control.

## DEVELOPMENT OF AN EFFICIENT MULTI-SPECIES DETECTION DEVICE FOR TARGETED MOP UP

At present, possum populations are monitored mainly using leg-hold traps, following a nationally standardised protocol (NPCA 2004), whereas rats and house mice (and stoats to some extent) are monitored by either snap-traps or tracking tunnels (King 1983, Brown et al. 1996). The possum-trap-catch method is relatively expensive because the traps have to be checked each day and the resulting index is insensitive, and therefore imprecise, at low density (Thomas et al. 2003). For all four species, the traps or tracking tunnels are bulky, limiting the numbers that can be deployed by one person in a day. To overcome some of these disadvantages, a variety of light-weight interference devices have been developed (NPCA 2005). These can be deployed in large numbers and can be left unchecked to accumulate interference data over multiple nights or even weeks or months, increasing sensitivity in terms of probability of recording a detection per observer-visit. We have adapted that principle in developing a small low-cost light-weight multi-species interference device, the 'ChewTrack Card', that we initially hoped would record interference rates for all four of the small mammal invasive species we are interested in.

The CTC consists of plasticised card (Coreboard) folded and nailed to trees with bait forced into the channel openings along the sides of the card and also placed inside the fold (Figure 3). In addition, tracking ink is applied as shown, so animals can be identified by both the bite marks and tracks they leave. Although still in development, CTCs have now been used both to monitor pest population reductions (as reported above) and to map the distribution of isolated survivors. Thus far, detection rates for stoats have been very low compared with those typically recorded using traps (King 1983, Murphy et al. 1999), suggesting CTC interference is a poor index of stoat abundance. In contrast, detection rates for possums and rodents are far higher (Table 2). In one field trial in which house mouse snap-back trapping and CTC surveys were conducted simultaneously, we recorded mouse trapping rates of 2.6% while the CTC interference rates were 47%. CTCs appear likely to be useful for identifying any generalist omnivore species, as we have also incidentally detected hedgehogs (*Erinaceus europaeus*, Table 2).

The efficiency of the devices is best illustrated by possum data from the Hauhungaroa Range, central North Island, which also highlights how we envisage the tool being used to target mop-up control. In that area, 82,900 ha was aerially poisoned in winter 2005 in an intensive high-cost

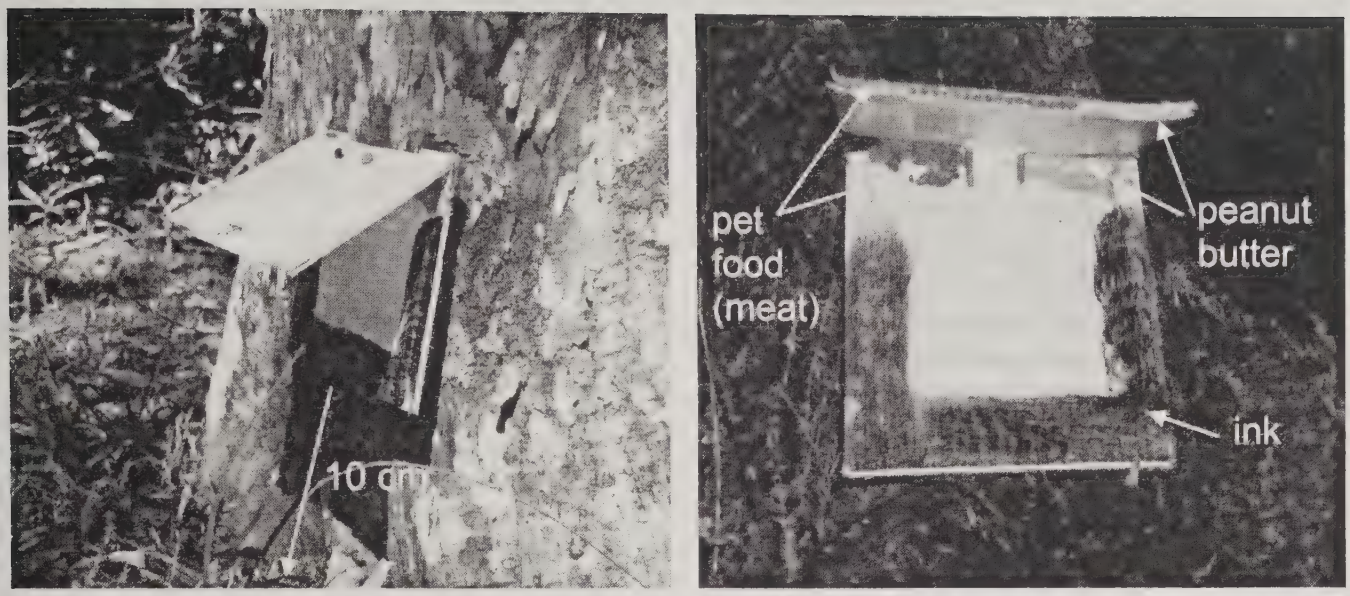



**Figure 3.** ChewTrack Card design, showing (a) side view, and (b) the location of tracking ink and bait (peanut butter for possums and rodents, meat paste for rodents, stoats and other carnivores).

**Table 2.** Small mammals detected during post-control trapping surveys in the Hauhangaroa Range, September 2005 to March 2006. The possum-specific trapping survey was conducted immediately after aerial poisoning and involved the setting and checking for three consecutive days of 88 randomly located lines of 10 traps spaced 20 m apart. The CTC survey involved setting 3070 CTCs 50 m apart along continuous transects systematically spaced 1 km part and checking them one week later.

| | Trap | CTC | | | | |
|---|---|---|---|---|---|---|
| | Possum | Possum | Rat | Mouse | Stoat | Hedgehog |
| Detections | 1 | 28[1] | 2025 | 1351 | 8 | 73 |
| % detections per device-day | 0.04 | 0.13 | 9.42 | 6.29 | 0.04 | 0.34 |
| % detections per field-day | 0.006 | 0.275 | 19.853 | 13.245 | 0.078 | 0.716 |

[1] Number of foci of possum detection, each focus comprised up to three consecutive chewed cards.

operation specifically designed to deliver near-total knockdown. That goal was successfully achieved (Table 1), and we have subsequently mapped pest survival and recovery in part of the area using CTCs.

Operational monitoring of that part of the area comprised 880 leg-hold traps set (and checked daily for 3 days) along 18 km of transect. This required 176 days of field effort and detected a single possum (i.e., 0.0004 captures per trap-night). In contrast, we deployed 3,070 chew cards along 153 km of transect and checked them a week later. This required just 102 days and detected at least 28 possums (i.e., 0.0013 detections per CTC night). More importantly, the CTCs were 46 times more efficient in terms of the number of possum detections per field day. The CTC surveys were conducted several months after the trapping survey, so some of this greater sensitivity is likely to reflect an increase in possum detectability, which is known to increase two- or three-fold with increasing time since poisoning (Forsyth et al. 2003), but the

difference in detections per day is far greater than that, so we are confident the efficiency gain is real.

Both the trapping and CTC surveys confirm extremely low numbers of survivors. A mean distance between capture and recapture of about 160 m has been recorded elsewhere in the Hauhungaroa Range (Morgan et al. 2007), so the 153 km of CTC transects monitored equates roughly to 2,500 ha of possum range surveyed. Subsequent trapping has indicated there was usually just one possum at each focus of CTC detections, suggesting that there was likely to be only one female per 150-200 ha. Possum home ranges are typically only a few hectares and females produce only one or two young per year, so we assume reaggregation of males and females into reproductive clusters will result in isolated foci of survivors that gradually increase in numbers. One concept for mop-up control is therefore to conduct annual CTC surveys along transects spaced 1 km apart, with transect location shifted laterally by 0.25 km each year. Over a 4-year cycle, this should identify the location of almost all of the low-number isolated reproductive foci of possums and enable them to very efficiently and cost-effectively targeted, perhaps simply by placement of long-life toxic baits (Morgan 2005) at the few possum detection sites when the CTCs are checked.

## CONCLUSION

Local elimination of invasive, small mammals now appears to be technically feasible as a new strategy for pest control in large unfenced mainland areas of New Zealand, at least for slow-breeding possums. This achievability mainly reflects the sustained incremental improvement in aerial poisoning over several decades to the point where possum populations can now sometimes be reduced to small numbers of geographically isolated foci. Provided those foci can be located cheaply, this enables highly-targeted mop-up control of only a small fraction of the area, rather than the blanket whole-area coverage that is currently the norm for repeat control. Transforming the concept into reality will require greater consistency in the achievement of near-total knockdown, and further development and refinement of the perimeter-control and mop-up tools outlined in this paper, but we see no major or fundamental obstacles to achieving that result.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

Brown, K. P., H. Moller, J. Innes, and N. Alterio. 1996. Calibration of tunnel tracking rates to estimate relative abundance of ship rats (*Rattus rattus*) and mice (*Mus musculus*) in a New Zealand forest. New Zealand Journal of Ecology 20:271-275.

Coleman, J., and P. Caley. 2000. Possums as a reservoir of bovine tuberculosis. Pages 92-104 *in* T. L. Montague, editor. The brushtail possum: biology, impact and management of an introduced marsupial. Manaaki Whenua Press, Lincoln, New Zealand.

Coleman, J. D., K. W. Fraser, and G. Nugent. 2007. Costs and benefits of prefeeding for possum control. New Zealand Journal of Zoology 34:185-193.

Forsyth, D., R. Webster, B. Warburton, G. Nugent, M. C. Coleman, C. Thomson, and W. A. Link. 2003. Effect of habitat, season, trap shyness and timing on RTCI estimates. Landcare Research Contract Report LC0203/001 (unpublished).

King, C. M. 1983. The relationships between beech (*Nothofagus* sp.) seedfall and populations of mice (*Mus musculus*) and the demographic and dietary responses of stoats (*Mustela erminea*), in three New Zealand forests. Journal of Animal Ecology 52:141-166.

King, C. M., and E. C. Murphy. 2005. Stoat. Pages 261-286 *in* C. M. King, editor. The handbook of New Zealand mammals. Oxford University Press, Melbourne, Australia.

Innes, J. 2005. Ship rat. Pages 187-203 *in* C. M. King, editor. The handbook of New Zealand mammals. Oxford University Press, Melbourne, Australia.

Innes, J., and G. Barker. 1999. Ecological consequences of toxin use for mammalian pest control in New Zealand – an overview. New Zealand Journal of Ecology 23:111-128.

Morgan, D.R. 1993. Multi-species control by aerial baiting: a realistic goal? New Zealand Journal of Zoology 20:367-372.

Morgan, D.R. 2004. Maximising the effectiveness of aerial 1080 control of possums (*Trichosurus vulpecula*). Unpublished PhD thesis. Lincoln University, Lincoln, New Zealand.

Morgan, D.R. 2005. Long-life baits for enhancing maintenance control of possum populations. Advances in Vertebrate Pest Management IV:193-210.

Morgan, D. M., and G. Hickling. 2000. Techniques used for poisoning possums. Pages 143–153 *in* T. L. Montague, editor. The brushtail possum: biology,

impact and management of an introduced marsupial. Manaaki Whenua Press, Lincoln, New Zealand.

MORGAN, D. R., G. NUGENT, AND B. WARBURTON. 2006. Benefits and feasibility of local elimination of possum populations. Wildlife Research 33:605-614.

MORGAN, D. R., G. NUGENT, D. GLEESON, AND R. HOWITT. 2007. Are some possums untrappable, unpoisonable, and therefore unmonitorable? Landcare Research Contract Report LC0607/143 (unpublished).

MURPHY, E., AND P. BRADFIELD. 1992. Change in diet of stoats following poisoning of rats in a New Zealand forest. New Zealand Journal of Ecology 16:137-140.

MURPHY, E., L. ROBBINS, J. B. YOUNG, AND J. E. DOWDING. 1999. Secondary poisoning of stoats after an aerial 1080 operation in Pureora Forest, New Zealand. New Zealand Journal of Ecology 23:175-182.

NPCA 2004. Protocol for possum population monitoring using the trap-catch method. National Possum Control Agencies, Wellington, New Zealand.

NPCA 2005. Protocol for low level possum abundance monitoring using the WaxTag method. National Possum Control Agencies, Wellington, New Zealand.

SWEETAPPLE, P. J., G. NUGENT, N. POUTU, AND P. HORTON. 2006. Effect of reduced possum density on rodent and stoat abundance in podocarp-hardwood forests. DOC Research & Development Series 231. Department of Conservation, Wellington, New Zealand.

THOMAS, M. D., J. A. BROWN, F. W. MADDIGAN, AND L. A. SESSIONS. 2003. Comparison of trap-catch and bait interference methods of estimating possum densities. New Zealand Plant Protection 56:81-85.

WILSON, D., G. MCELREA, R. HEYWARD, AND C. THOMSON. 2006. Weta and mice – prey and predators in an alpine habitat. Kararehe Kino – Vertebrate Pest Research 9:6-8.

# MULTIPLE-SPECIES EXCLUSION FENCING AND TECHNOLOGY FOR MAINLAND SITES

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***Abstract:*** Eradication of invasive vertebrate pests from increasingly large islands has become an important wildlife management and conservation tool internationally. Success on islands has prompted attempts to exclude and eradicate vertebrate pests from mainland sites. Early mainland exclusion efforts often failed due to ineffective or poorly maintained barriers to pest reinvasion. Over the last 10 years, we have conducted extensive experiments to design effective pest exclusion technology. We have determined the behaviour and physical abilities of many of the vertebrate pest species found in New Zealand and other parts of the world. Pest species have been tested against a variety of fence designs with the aim of developing 100% effective barriers. We found that fences which relied on the use of electrified wires proved ineffective for most species, whereas barriers that exceeded the physical capability of the target pests were reliable. Two multi-species fence designs excluded every pest tested. The designs excluded rodents (including mice), lagomorphs, mustelids, hedgehogs, brushtail possums, cats, dogs, feral pigs, goats, deer, Javan macaque and domestic livestock. The outcome of this research programme has been the commercial availability of two designs of Xcluder™ pest proof fence. Supporting components and technology, such as pest-free pedestrian and vehicle gates, waterway gates and remote surveillance systems to mitigate reinvasion risks have enabled projects to succeed. Over 20 exclusion barrier systems have now been constructed in areas up to 3,400 ha in size and have allowed multi-species eradication attempts. With the successful removal of vertebrate pests, many projects are now undertaking significant restoration programmes including the reintroduction of threatened wildlife species to mainland sites.

***Key Words:*** barrier, behaviour, eradication, exclusion, house mouse, invasive species, mustelid, pest proof fencing, rodent, Xcluder™

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## INTRODUCTION

Eradication of invasive vertebrate pests from increasingly large islands has become an important wildlife management and conservation tool internationally. Since the 1970s, rodents have been eradicated from an increasing number of islands around New Zealand and, more recently, elsewhere in the world (Veitch and Bell 1990, Taylor et al. 2000, Towns and Broome 2003). Success on these islands has now prompted attempts to exclude and eradicate multiple species of vertebrate pests from mainland sites (Speedy et al. 2007).

Exclusion fencing is being used internationally to protect areas of high conservation value or to create 'islands' of protected habitat for native fauna. It has proven a particularly valuable tool in aiding the reintroduction of threatened species to areas from which they have been previously eliminated or displaced by pests (e.g. Dufty et al. 1994, Moseby and O'Donnell 2003, Speedy et al. 2007). However, mainland pest exclusion and eradication relies on effective fence design to be a cost-effective and sustainable pest management strategy (Clapperton and Day 2001).

The design of an exclusion fence must be based on the behaviour and physical abilities of the animals it aims to exclude. Many historical exclusion fences were not experimentally tested (Long and Robley 2004), were focused on exclusion of single rather than multiple species (Aviss and Roberts 1994) and often failed because of faulty design, poor construction, or lack of maintenance (e.g., Day and Flight 2002). Often, the process of fence development has been undertaken by independent organisations and individuals around the world, leading to many fence designs, with varied success, for a diverse range of species and situations. Filling knowledge gaps about pest animal behaviour and physical abilities would allow development of optimal, cost-effective fence designs (Long and Robley 2004).

Fences that rely upon the responses of animals to electric wires have been used extensively overseas for pest control (McKillop and Sibly 1988). Electric fences are primarily psychological rather than physical barriers and are effective against some mammal pests, e.g. rabbits (*Oryctolagus cuniculus cuniculus*) (McKillop et al. 1993), and fox (*Vulpes vulpes*) (Minsky 1980). However, brushtail possums (*Trichosurus vulpecula*) quickly breach an electric fence during a power failure (Cowan and Rhodes 1992, Clapperton and Matthews 1996). Stoats (*Mustela erminea*) can move so quickly up a fence that they can pass the electrified wires between energy pulses (Day and MacGibbon 2002). When rodents, possums and cats (*Felis* spp.) are sufficiently motivated electrified wires do not prevent either species from crossing fences (Clapperton and Matthews 1996, Day and Flight 2002).

Physical barrier fences that exceed the behaviour and physical abilities of the target pest offer a much greater chance for effective exclusion. Barrier fences have been developed for single species, but have often still incorporated electric wires into the physical barrier design (e.g., fox, (Poole and McKillop 2002) and dingo (*Canis lupus dingo*, Bird et al. 1997)). Barrier fences have rarely been designed to exclude the entire suite of pests present at a site. In many cases, project managers have been resigned from the beginning to the fact that their fences will only contain a proportion of the target animals (Long and Robley 2004). One of the few groups that have experimentally evaluated barrier fence type designs for multiple pest species is the Karori Wildlife Sanctuary (Karori Wildlife Sanctuary Trust Inc. 1998). By measuring the physical abilities of the target species (e.g. maximum jump height, climbing ability etc.) they developed an effective barrier for all target pests except mice (Karori Wildlife Sanctuary Trust Inc. 2001).

This paper describes outcomes from over 10 years of research to design cost effective fences that are completely effective for multiple assemblages of pest species. The research started in 1996 (as a result of landowner desire to exclude herbivore animal pests from native plantings), and was initially focused on pests found in New Zealand. Research has since been extended to Australia, Hawaii and Mauritius. The entire focus of the research described in this paper has been to challenge the notion that no fence is likely to be 100% effective for 100% of the pests 100% of the time (Aviss and Roberts 1994, Coman and McCutchan 1994). This was achieved by designing and experimentally determining the efficacy of practical effective fences for ALL pest species present at a site. However, even the most effective fence design will only continue to be effective if it is regularly monitored for reinvasion risks and is well maintained (Sexton 1984, Coman and McCutchan 1994, Day and Flight 2002). Therefore, as the experimentally successful fence designs described in this paper have been built in the field (18 sites), in-situ analysis of their long term pest exclusion efficacy has been made and is described.

## METHODS

### Fence Designs Tested

Three basic fence designs were experimentally evaluated for their efficacy to contain or exclude pests. The main designs tested were the electric fence, the Xcluder™ "Tui" fence and the Xcluder™ "Kiwi" fence (Figure 1).

The electric fence consisted of a wooden post and wire fence 1,200 mm high, with backing wires and wire mesh placed up its length. A 'skirt' of mesh was pinned to the ground and extended >300 mm horizontally out towards the pests. The mesh skirt was then covered with 50 mm of earth. For a description of how the wire mesh was chosen for these experiments see the "Wire mesh experiments" section below. Two 300 mm long steel outriggers were placed on the fence: one at the top, angled slightly upward, and one at 600 mm above the ground on the same angle. Each outrigger had an identical configuration of 5 wires (3 electrified wires and 2 ground wires) running parallel along the fence length. The electric wires were powered by a Gallagher fence energiser with an output of 58 pulses/sec at 8,500 volts per pulse. The inside electric wire was within 20 mm of the fence mesh and the outside wire was 300 mm from the vertical face of the mesh. Although several minor modifications were made to the design early in the research, this fence design remained similar to that previously described by Clapperton and Matthews (1996) as a brushtail possum barrier.

The Xcluder™ "Tui" fence was designed after the electric fence efficacy trials had been completed and we had learned something about the behaviour of the pests we wished to exclude. The Tui fence consisted of a base fence made of 1200 mm high wooden posts, with hi-tensile backing wires and wooden battens. The base

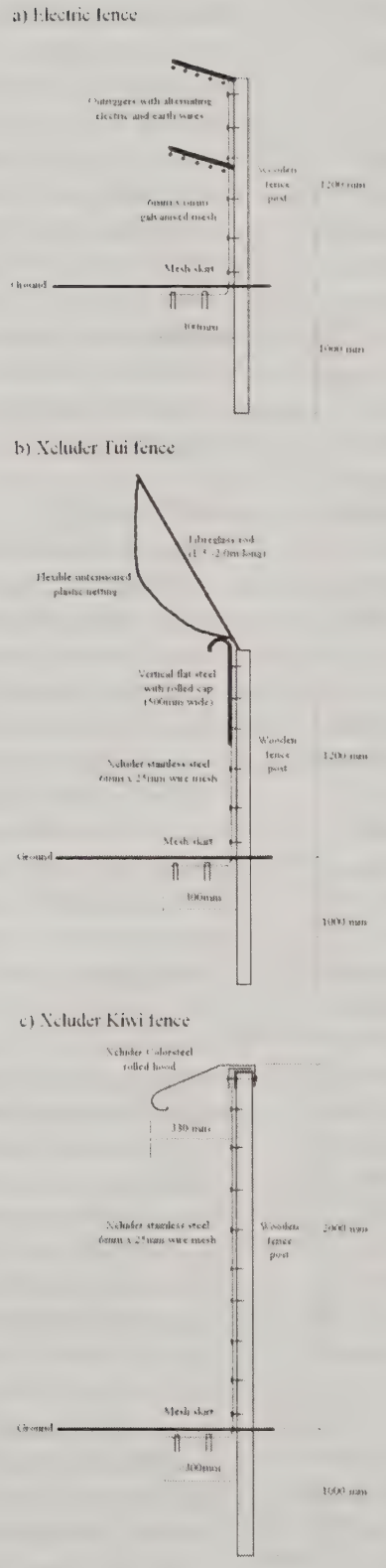



**Figure 1.** Configuration of the three main fence designs experimentally evaluated for their pest containment efficacy: a) electric fence; b) Xcluder™ "Tui" fence; and c) Xcluder™ "Kiwi" fence.

fence was very similar to a standard 9-wire post and batten livestock fence, commonly used in New Zealand. Attached to the base fence was wire mesh and a wire mesh skirt similar to that used on the electric fence. The wire mesh extended 800 mm up the face of the fence. A 500 mm wide flat vertical sheet of steel (Colorsteel®) was placed on to the top portion of the fence, overlapping the mesh at the bottom and extending to a height of 1,300 mm above the ground. An 80 mm wide half-circle 'cap' facing towards the pests was manufactured into the top of the flat steel sheet. Above the flat steel sheet a 1.5-2.0 m high section of flexible plastic woven horticultural 'bird' netting was suspended in a loosely tensioned fashion on supple fibreglass rods. The fibreglass rods were mounted in the fence so that they leaned slightly toward the pests, creating a sag in the plastic netting. The design of the fence was unique in that the bottom portion was sturdy and rigid, while the top portion was deliberately flexible and able to move freely in the wind or when animals climbed on it. Several small modifications to this design were made during the research process.

The Xcluder™ "Kiwi" fence design was built after our animal behaviour observations had been completed for both the electric fence and the Xcluder™ "Tui" fence, so it was designed to defeat all of the pest escape behaviours we had already observed. The fence consisted of a 2 m high base fence of wooden posts, backing wires and wooden battens. Wire mesh was affixed up the entire length of the base fence and a >300 mm wide mesh skirt facing the pests was pinned to the ground and covered. At the top of the fence a sheet of 600 mm wide steel (Colorsteel®) was folded and rolled to form a 'hood' that was mounted at the top of the fence and extended 330 mm horizontally towards the pests. The steel hood was mounted on custom-built brackets, so that it was sturdy and would not move when animals climbed or jumped on it.

For the experimental evaluation of fence efficacy with Hawaiian and Mauritian species and conditions, the Xcluder™ Kiwi fence was modified slightly. For the Hawaiian research, the wire mesh skirt at the base of the fence was modified to be fixed to lava substrates with a cement-based mix (see Burgett et al. 2007 for details). In Mauritius, the shape and length of the Xcluder fence hood was extended vertically to counter the extra reach of Javan macaque (shape and exact design of modified hood described in Day 2004).

**Pest Species and Locations**

Sixteen pest species were used during the course of our animal behaviour and fence efficacy experiments. The pest species used, the locations in which the trials were conducted for each species and the number of animals of each species tested are described in Table 1. The number of animals used in the experiments was variable for each species and fence design for three reasons. Firstly, some species were of particular interest because of their perceived better escape ability (e.g., mice [*Mus musculus*], ship rats [*Rattus rattus*], possums [*Pseudocheirus peregrinus*], and cats [*Felis catus*]), so the numbers tested was higher. Secondly, some species were difficult to capture, handle or test in an experimental situation, so we were forced to accept lower numbers of individuals (e.g., stoats [*Mustela erminea*], hares [*Lepus europaeus occidentalis*]). Thirdly, the experiments were conducted on the basis that once satisfied a fence had failed to contain a pest species, research with that fence design was discontinued for all species. Because of the variable animal numbers, the in-situ fence efficacy data (described below) is of greater importance in proving the efficacy of the designs for some species.

The animals used in the fence efficacy and animal behaviour experiments were caught in the wild using live-capture box traps and were transported to the experimental facility and tested within 24 hrs of capture. Animals were provided with food, water and shelter during their time in captivity. At the conclusion of experiments pest animals were humanely euthanized (as it is illegal and considered unethical to release pest animals back into the wild in New Zealand). All animal experiments were conducted with appropriate Animal Ethics Committee approval and permits for each location.

**Wire Mesh Experiments**

As the aim of this research project was to design fences that were effective for ALL vertebrate pests, the research began by determining the size and aperture of wire mesh required to contain the smallest of the target pests. In New Zealand, mice were the smallest target species and this later proved also to be the case in Hawaii and Mauritius. Therefore, the largest aperture of mesh required to prevent all independent juvenile mice from passing through a pest fence was considered the minimum standard required for construction of a total pest exclusion barrier.

**Table 1**. Species tested and numbers used for all animal behaviour and fence efficacy experiments, plus species not experimentally tested but present outside in-situ Xcluder™ Tui and Xcluder™ Kiwi fences (see Table 5 for in-situ sites).

| Species | Number tested | Locations tested | Outside in-situ fences |
|---|---|---|---|
| House mouse (*Mus musculus*) | 220 | NZ, H, M | ✓ |
| Ship rat/Black rat (*Rattus rattus*) | 108 | NZ, H, M | ✓ |
| Norway rat (*Rattus norvegicus*) | 33 | NZ | ✓ |
| Ferret (*Mustela furo*) | 14 | NZ | ✓ |
| Stoat (*Mustela erminea*) | 6 | NZ | ✓ |
| Hedgehog (*Erinaceus europaeus occidentalis*) | 10 | NZ | ✓ |
| Rabbit (*Oryctolagus cuniculus cuniculus*) | 22 | NZ | ✓ |
| Hare (*Lepus europaeus occidentalis*) | 7 | NZ, M | ✓ |
| Brushtail possum (*Trichosurus vulpecula*) | 87 | NZ | ✓ |
| Cat (*Felis* sp.) | 139 | NZ | ✓ |
| Pig (*Sus scrofa*) | 11 | NZ, H, M | ✓ |
| Indian mongoose (*Herpestes javanicus*) | 32 | H, M | ✓ |
| Mouflon sheep (hybrid) (*Ovis musimon*) | 12 | H | |
| Indian house shrew (*Suncus murinus*) | 12 | M | ✓ |
| Javan macaque (*Macaca fascicularis*) | 42 | M | ✓ |
| Dog (*Canis familiaris*) | 11 | NZ, M | ✓ |
| Javan deer (*Cervus timorensis*) | | | ✓ |
| Pacific rat (*Rattus exulans*) | | | ✓ |
| Weasel (*Mustela nivalis vulgaris*) | | | ✓ |
| Goat (*Capra hircus*) | | | ✓ |
| Fallow deer (*Dama dama*) | | | ✓ |
| Red deer (*Cervus elephus*) | | | ✓ |
| White-tailed deer (*Odocoileus virginianus borealis*) | | | ✓ |
| Guttural toad (*Bufo guttularis*) | | | ✓ |
| Tenrec (*Tenrec ecaudatus*) | | | ✓ |

NZ = New Zealand; H = Hawaii; M = Mauritius.

Wild mice were captured in live traps and housed in social groups in standard pet cages to form a breeding population. Once breeding was regularly producing juveniles, mice of known ages and sizes were placed in an experimental box to test what type and aperture of wire mesh they could pass through. The experimental box consisted of two chambers separated by a section of the wire

mesh to be evaluated. Mice were placed on one side of the mesh and food, water and shelter were placed on the other side. The ability of mice to pass through the mesh was recorded for periods of up to 24 hrs. Different sizes and shapes of commercially available wire mesh (ranging from 25 mm aperture down to 4.4 mm aperture) were used to determine the maximum aperture that could be considered mouse proof.

In addition to wire mesh tests conducted specifically with mice, we also evaluated the size of mesh required to contain juveniles of most other species during our fence efficacy experiments.

**Fence Efficacy and Animal Behaviour**

A series of experimental facilities were constructed at the three locations in which we conducted our fence research (Cambridge, New Zealand; Kona, Hawaii; Mauritius). At each location, we built one or more experimental enclosures. Each enclosure was constructed with one of the pest proof fence designs facing into the enclosure around its perimeter. The enclosures were octagonal in shape and approximately 12 m across their width: enclosures in New Zealand and Mauritius were constructed on open areas of short mown grass, while the Hawaii enclosures were built on an old lava flow. For the smaller species, such as mice, ship rats, Norway rats (*Rattus norvegicus*), and house shrew (*Suncus murinus*), a much smaller enclosure of approximately 4 $m^2$ was used so that the animals could be physically observed during the experiment. A 2 m high covered observation tower was built beside the enclosures to allow observation and video recording of animal behaviour and fence interaction when pests were placed inside the enclosure in an "escape test".

The escape test was used to determine the efficacy of different fence designs. Individual wild-caught pest animals were placed into one of the enclosures and observed from the observation tower for escape behaviours. Because the animals were wild, all exhibited motivation to escape from the barren enclosures. Observations for each animal focused on the pushing, digging, climbing, jumping and chewing abilities of each species. Video cameras were used to record all escape attempts, so that we could analyse in detail the escape behaviour and method. In addition, we made physical measurements of jumping heights and distances made by each animal.

Pest animals were tested under three different levels of motivation to escape: (1) animals were introduced to the enclosure and left with food, water and shelter to explore and escape without any human presence for up to 3 weeks (low pressure); (2) animals were introduced to the enclosure and were observed from the observation tower for at least the first 3 hours of the escape test (medium pressure); or (3) animals were introduced to the enclosure and the animal handler remained in the enclosure with the pest animal as it tried to escape (high pressure). Using this range of test situations, animals exhibited their full range of behaviour and physical abilities, from planned, calculated and methodical exploration of the enclosures, to vigorous and rapid physical escape attempts using their maximum physical abilities.

**In-situ Efficacy of Fences**

At the conclusion of our initial experimental research, a number of conservation groups chose to build either Xcluder™ "Tui" or Xcluder™ "Kiwi" fences around high value conservation areas. For all sites fenced with either fence design (and where the sites were already pest free or total pest eradication attempts have been undertaken; see Speedy et al. 2007 this proceedings for a summary of some of the sites), data on the long-term exclusion efficacy of the fences was collected as part of project management. At each of these sites various assemblages of pest species were present immediately outside the fences. These species posed immediate potential for reinvasion if fence designs were not satisfactory and ongoing potential for reinvasion if the integrity of the fences were compromised by human error, fence damage (e.g., tree fall, flooding damage), fence component failure, or malicious activity. Therefore, data collected from the in-situ sites, where pest reinvasion potential is continuous over extended periods, is considered to be the 'ultimate' measure of the long term efficacy of: (1) the exclusion fence designs; (2) all of the associated fence components required to make a pest-proof enclosure in a real site (e.g. vehicle and pedestrian gates, water gates etc); and (3) the ability of each project to manage reinvasion risk at their site.

At the in-situ sites, potential invaders included mixed assemblages of the 16 species tested experimentally, plus at least nine species that have not been experimentally tested for exclusion efficacy (Table 1). Data recorded at each in-situ site included a full description of the fence and it's associated components, proof of the presence of each pest species outside the fence, details and efficacy of eradication attempts inside the fence,

details of any potential compromises to the integrity of the pest-proof fence over time, and records of any pest invasions into the fenced area and the outcome of the invasion. The data collected has been used in this paper to summarise the in-situ efficacy for fences at 18 sites.

## RESULTS

### Wire Mesh Experiments

Using different sizes of commercially available wire mesh, we determined that 100% of juvenile (but independent and mobile) mice were able to pass through welded mesh with a hole size of 10 x 10 mm, and one juvenile mouse passed through 8 x 8 mm aperture mesh (Table 2). The smallest aperture through which any mouse passed was a hole size of 7.1 x 40 mm. Adult mice were larger and therefore were restricted by mesh smaller than 10 x 10 mm, but 71% of adults passed through the 10 x 10 mm mesh (Table 2).

House shrew juveniles were also able to pass through 10 x 10 mm mesh. All other animal species tested were contained by mesh of 13 mm aperture or greater.

Because this research, programme aimed to design effective fences for ALL pest animals, mesh with an aperture of no more than 6 mm in one dimension (to provide a safety margin) was used on each fence design for all subsequent research. On the experimental Xcluder™ "Tui" and Xcluder™ "Kiwi" trial fences, 6 x 25 mm aperture 316 grade stainless steel mesh was used, and no pest animal ever passed through this mesh during our experiments.

### Fence Efficacy and Animal Behaviour

Table 3 describes the number of animals of each species tested and the percentage that were contained in the escape test by the three main fence designs. The predominant escape behaviours exhibited by each species and their

**Table 2.** Summary of escapes made by mice, the smallest pest species tested in experiments, when placed in small cages made of various sizes and types of wire mesh.

| Mesh hole size | | Adult | | Juvenile | |
|---|---|---|---|---|---|
| (Length mm x width mm) | Mesh type | N | % escape | N | % escape |
| 25 x 25 | Welded | 35 | 100 | 24 | 100 |
| 2 x 25 | Welded | 16 | 100 | 15 | 100 |
| 19 mm diamond | Chainlink | 23 | 100 | 15 | 100 |
| 13 mm hexagon | Welded | 23 | 100 | - | - |
| 12 x 12 | Welded | 38 | 100 | 24 | 100 |
| 12 mm diamond | Chainlink | 23 | 74 | 15 | 100 |
| 10 x 10 | Welded | 26 | 71 | 15 | 100 |
| 8 x 8 | Welded | 26 | 0 | 24 | 4* |
| 6 x 40 | Woven | 38 | 0 | 33 | 0 |
| 6 x 32 | Woven | 29 | 0 | 24 | 0 |
| 6 x 25 | Welded | 29 | 0 | 24 | 0 |
| 6 x 12 | Welded | 29 | 0 | 24 | 0 |
| 6 x 6 | Welded | 29 | 0 | 17 | 0 |
| 5.3 x 24.3 | Welded | 68 | 0 | 43 | 0 |
| 4.4 x 40 | Woven | 35 | 0 | 26 | 0 |

* Smallest mesh hole through which a juvenile mouse passed was 7.1 x 40 mm.

**Table 3.** Number of pests of each species tested with the three main fence designs and the percentage of each species that were contained by each fence design.

| | Electric fence | | Xcluder™ Tui | | Xcluder™ Kiwi | |
|---|---|---|---|---|---|---|
| **Species** | **N** | **% contained** | **N** | **% contained** | **N** | **% contained** |
| Mouse | 13 | 61 | 30 | 100 | 167 | 100 [1] |
| Ship rat | 5 | 40 | 22 | 100 | 87 | 100 |
| Norway rat | 6 | 100 | 17 | 100 | 16 | 100 |
| Ferret | 3 | 100 | 3 | 100 | 8 | 100 |
| Stoat | 4 | 25 | 4 | 100 | 2 | 100 |
| Hedgehog | 3 | 100 | 4 | 100 | 3 | 100 |
| Rabbit | 3 | 100 | 15 | 100 | 11 | 100 |
| Hare | 2 | 100 | 2 | 100 | 5 | 100 |
| Possum | 25 | 52 | 30 | 100 | 42 | 100 |
| Cat | 13 | 8 | 45 | 100 | 58 | 100 |
| Pig | | | | | 11 | 100 |
| Mongoose | | | | | 32 | 100 |
| Mouflon sheep | | | | | 12 | 100 [2] |
| House shrew | | | | | 12 | 100 |
| Javan macaque | | | | | 42 | 100 [3] |
| Dog | | | 3 | 100 [4] | 11 | 100 [4] |

[1] For mice tested on lava in Hawaii, the Xcluder™ Kiwi fence skirt had to be modified to be 100% effective (Burgett et al. 2007).
[2] Mouflon sheep (hybrids) were tested by D. Goltz at separate sheep-fence test facility (Burgett et al. 2007).
[3] For Javan macaque, the shape and length of the Xcluder™ Kiwi hood was lengthened.
[4] Dog numbers include both domestic working dogs (NZ) and feral dogs (Mauritius).

associated physical capabilities are summarised in Table 4. The electric fence effectively contained all tested Norway rats, ferrets, hedgehogs, rabbits and hares (although the number of individuals of each species tested was low). Behaviourally, these species attempted to push through the fence at the base or tried to dig under. Animals systematically patrolled the length of the experimental enclosure fences, appearing to search for perceived weak points and trying to push through gaps in the mesh. Pushing was often the first and most common escape behaviour exhibited, especially by non-climbing pest species. However, none of the pests tested were able to push though the mesh or make any significant impacts on the wire mesh to create holes. Pushing behaviour did not result in any escapes for any pests on any of the fence designs.

Norway rats, rabbits, hares and stoats all dug directly at the base of the fence and, on encountering the horizontal mesh skirt below the surface, began digging further out from the fence to a maximum distance of 200 mm. These animals then began digging against the fence in a new position. None of the digging animals chose to start digging more than 250 mm from the fence and none found the leading edge of the mesh or were able to dig under the skirt (even when housed in the enclosures for as long as 3 weeks). None of

**Table 4.** Summary of behavioural responses and physical abilities of pest species when trying to cross various pest proof fence designs and components.

| Species | Push | Dig | Climb | Jump height (mm) | Chew | Mesh hole size to contain (mm) | Contained by electric wire | Learn |
|---|---|---|---|---|---|---|---|---|
| Mouse | ✓ [1] | ✓ | ✓✓ [2] | 400 | ✓ | 6 | X [3] | ✓ |
| Ship rat | ✓ | ✓ | ✓✓ | 800 | ✓ | 13 | X | ✓ |
| Norway rat | ✓ | ✓✓ | ✓ | 800 | ✓ | 13 | ✓ | ✓ |
| Ferret | ✓ | X | ✓ | X | ✓ | 13+ | ✓ | ✓ |
| Stoat | ✓ | ✓✓ | ✓✓ | 1,200+ | ✓ | 13 | X | ✓ |
| Hedgehog | ✓✓ | X | X | X | X | 50 | ✓ | ✓ |
| Rabbit | ✓ | ✓✓ | X | X | ✓ | 50 | ✓ | ✓ |
| Hare | ✓ | ✓✓ | X | 800 | ✓ | 50 | ✓ | ✓ |
| Possum | ✓ | X | ✓✓ | 1,200+ | ✓ | 50 | X | ✓ |
| Cat | ✓✓ | X | ✓✓ | 1,800+ | ✓ | 50 | X | ✓ |
| Pig | ✓✓ | ✓ | X | X | ✓ | 100 | NT [4] | ✓ |
| Mongoose | ✓ | X | ✓✓ | X | ✓ | 50 | NT | ✓ |
| Mouflon sheep | ✓✓ | X | X | 1,800+ | X | 100 | NT | ✓ |
| House shrew | ✓✓ | ✓ | ✓✓ | 300 | ✓ | 10 | NT | ✓ |
| Javan macaque | ✓ | X | ✓✓ | <1,800 | X | 100 | NT | ✓ |
| Dog | ✓ | ✓✓ | X | 1,800+ | ✓ | 100 | NT | ✓ |

[1] ✓ = exhibits behaviour regularly and competently.
[2] ✓✓ = excels at behaviour and uses very frequently during escape attempts.
[3] X = does not usually exhibit behaviour.
[4] NT = Not tested.

the species contained by the electric fence chose to climb the fence repeatedly, and therefore they did not significantly challenge the electrified outriggers: none of these animals made any attempts to climb around the outside of the outriggers and all appeared to be repelled by the first shock they received when investigating the wire.

The electric fence did not contain mice, ship rats, stoats, brushtail possums or cats, who after trying to push through the fence, or digging at the base, all attempted to escape by climbing and jumping. Mice were able to climb or jump up the wire mesh and 39% of those tested passed between the mesh and the inside electric wire (20 mm gap) without receiving a shock. Those that received a shock were knocked to the ground. Ship rats ran

and jumped up the fence mesh and 60% climbed under, through or around the electric wire outriggers. On occasions, they received shocks that knocked them to the ground, but this did not prevent the persistent animals from passing the wires. Stoats ran up the mesh on the fence and through the electric wires extremely quickly: 3 individuals never received shocks from the wires, as they passed over them between pulses (~1 sec apart). The one stoat that received a shock did not subsequently cross the fence. Possums and cats climbed and jumped at the electric fence regularly without being under any pressure to do so. While 52% of possums and 8% of cats were contained by the electric fence, most continued to attempt to cross the fence by climbing and jumping despite receiving multiple shocks. One possum received 42 shocks before finally crossing around the outside of the outriggers. Both possums and cats were able to jump to the top of the top outrigger directly from the ground.

The Xcluder™ "Tui" fence contained all pest animals that were tested against it (Table 3). The mesh and mesh skirt at the base of the fence functioned exactly the same in these experiments as has been described above for the pushing and digging behaviour of animals during the electric fence trials. No pest animals were able to push through the mesh or dig under the skirt.

The flat sheet of smooth steel with a rolled cap at the top prevented all species except cats from climbing up the fence. Animals climbed the mesh to the base of the steel and then reached, scratched and jumped at the steel to try and move forward. The 500 mm wide sheet with cap provided no foot-holds and was too wide for all animals except cats to reach or jump across. One exception to this pattern of behaviour was observed in earlier research. Stoats were able to jump across a 600 mm wide sheet of flat steel from the wire mesh just below the sheet (T. Day, unpublished data). This ability prompted the use of a rolled cap at the top of the sheet that was used on the Xcluder™ "Tui" fence design described here. Positioning the top of the sheet 1,300 mm above the ground meant that cats were the only species able to jump above it directly from the ground (see jumping heights for individual species in Table 4).

The flexible plastic netting suspended on fibreglass rods contained all cats that jumped at and attempted to climb it. The unstable nature of the netting did not allow any cats sufficient grip to climb to the top and over it. For larger cats their body weight caused the netting to collapse toward the ground until the cats hind legs touched the ground. When this occurred all cats let go of the netting. In earlier trials using similar netting at the top of a fence, two large male cats were able to climb up and over a 900 mm high section of the same netting suspended on rods.

The Xcluder™ "Kiwi" fence also contained all pest animals that were experimentally tested against it (Table 3). Again the mesh and mesh skirt functioned in the same manner as we had previously observed for the electric and Xcluder™ "Tui" fences. Pest animals readily climbed to the top of the 2 m high mesh on the Kiwi fence, but no animals were able jump to the top of the fence directly from the ground (Table 4). As there were no gaps bigger than 6 mm anywhere in the fence, no animals were able to squeeze through the fence at any point. The 330 mm wide hood at the top of the Xcluder™ "Kiwi" fence forced climbing and jumping animals to reach or jump outwards and away from the fence in an attempt to move around the sheet. Javan macaque and cats had the longest reach around the hood (the hood was lengthened to 330 wide x 600 mm long for Javan macaque), but both species were unable to grip the smooth surface of the hood and could not pull themselves around to the top of the fence.

Several behavioural characteristics common to most pest species and all fence designs were observed during these experiments. Pest animals focused over 75% of all escape attempts at corners in the fence rather than on straight sections. Animals tended to run along the base of the fence and only attempt to dig, push, climb or jump over the fence when they encountered a change in fence direction. As such, the corners of the fence (especially the inside angles) received much more escape 'pressure' than the straight sections of the fence. Further, in our early research, stoats, possums and cats effectively used tight corners (<120°) to assist them to jump higher or further than we observed on straight sections of fence. For example, on more than one occasion, stoats, possums and cats all crossed 600 mm wide flat sheets of steel in 90° corners by jumping back and across the corner to the top of the opposing flat sheet.

Sequential analysis of the behaviour of individual animals during the escape test clearly showed evidence of animal learning, using a process of trial and error. Many animals would attempt a method of escape repeatedly until they appeared 'satisfied' they could not escape via that method. They would then modify their escape

behaviour and try again until they were either successful or modified their behaviour again. Most animals only stopped trying to escape from the fences when they appeared to have exhausted all potential avenues for escape and had displayed a full range of escape behaviours and physical abilities.

**In-situ Efficacy of Fences**

Data collected at the in-situ sites demonstrated that Xcluder™ "Tui" and Xcluder™ "Kiwi" fences can effectively exclude all target pest animals in the long term as long as the integrity of the fence has not been compromised (Table 5). Effective pest monitoring regimes inside all fenced sites revealed no evidence of pest animal incursion past either fence type without a specific fence risk event. This data supports the efficacy data collected experimentally. Several new pest species that were not tested experimentally, as well as the suite of pest species that were tested, were confirmed to be present immediately outside one or more of the in-situ or experimental fences (Table 1). None of these species were detected inside any of the fenced sites, except after known reinvasion risk events. No unexplainable pest animal detections (detections without an associated known compromise to the fence) were recorded at any of the in-situ sites.

Pest animal reinvasion events were recorded at nine of the 18 monitored sites (Table 5). After known compromises to fence integrity, mice, ship rats, brushtail possums, stoats, cats or white-tailed deer (*Odocoileus virginianus*)were all found inside pest-proof fenced areas immediately following the compromise. Reasons recorded for the fence integrity to be compromised in-situ included: (1) vehicle or pedestrian gates being left open; (2) human error (platform or vehicle being accidentally left close to the pest fence allowing animals to jump over); (3) erosion damage under the base of the fence; and (4) tree falls crushing the fence and leaving an opening. In two cases (Young Nicks Head and Macraes Flat) the reinvasion led to the re-establishment of mouse populations inside the fence. None of the other invasions resulted in any long-term pest presence inside the fence, so did not compromise the long-term pest-free goals of the sites. Only two sites with fence lengths greater than 1 km had no recorded invasion: Pitt Island (40 ha) and Maungatautari North enclosure (35 ha).

At Mt. Maungatautari, detailed records of every invasion risk event and the response to it has been recorded for the 47 km of Xcluder™ "Kiwi fence. At least 12 significant risk events (1 by vehicle gate open, 1 by water gate jammed open, 10 by tree falls) were recorded over a three year period (T. Day, unpublished data, P. de Monchy, personal communication). These risk events resulted in three recorded invasions: two events each resulting in a rat detection; one event resulting in a mouse detection. The detected invaders were removed in all three cases (as evidenced by animal capture and subsequent cessation of animal tracking). On all three occasions where invasion resulted, there was a significant time delay (between 6 and 24 hrs) between the fence compromise and staff being able to repair the breach. In contrast, when the remote surveillance system at Maungatautari was used, it enabled response in less than 3 hrs and no animal invasion was detected.

## DISCUSSION

This research has clearly demonstrated experimentally and in-situ that completely effective multi-species exclusion fence designs are possible. In an experimental situation, two of the fence designs we evaluated, the Xcluder™ "Tui" fence and the Xcluder™ "Kiwi" fence, excluded every individual from the 16 species we tested. In addition, nine further species found at the in-situ sites were excluded by one or other of the fence designs. While the number of individual animals tested experimentally was low for some species, the in-situ data provides excellent evidence of the long-term efficacy of the fence designs.

Mice, as the smallest of pests studied and encountered in our research, dictated the maximum mesh aperture and gap that could be allowed on any part of the fences if complete pest exclusion was to be achieved: to provide a small margin for error, there should be no gap on a fence bigger than 6 mm if mouse exclusion is desired. To achieve this tolerance in the field, precise construction techniques and exceptional product quality are required. On the in-situ fences, 6 x 25 mm aperture stainless steel welded wire mesh was used. This mesh provided the strength, consistent aperture and tolerance required to achieve success in the field. At Karori Wildlife Sanctuary, mice reinvaded because apertures on the woven mesh used did not consistently remain less than 6 mm (Karori Wildlife Sanctuary Inc. 2001).

As has been found by previous researchers (e.g., Minsky 1980, McKillop and Sibly 1988, McKillop et al. 1993), the electric fence design tested in this research effectively contained species with poor

**Table 5.** Detail of in-situ sites fenced with Xcluder™ "Tui" or Xcluder™ "Kiwi fence designs, the number pest species present outside the fence, evidence of pest exclusion efficacy and reinvasion events.

| Project (ha) | Fence design | Fence length (km) | Fence age (yrs) | Species outside/ excluded | Fence reinvasion events (reasons for reinvasion) and outcomes |
|---|---|---|---|---|---|
| Warrenheip (16) | Tui | 2.3 | 8 | 12 / 12 | Mice, rat, possum<br>**(gate open/erosion damage)**<br>all eradicated |
| Pitt Island (40) | Tui | 3.0 | 7 | 2 / 2 | No invasion |
| Rapanui Point (1) | Kiwi | 0.6 | 4 | 7 / 7 | No invasion |
| Lord Howe Island (<1) | Tui | 0.2 | 4 | 2 / 2 | No invasion |
| Mt Bruce (<1) | Kiwi | 0.22 | 4 | 11 / 11 | No invasion |
| Kiwi Encounter (<1) | Kiwi | 0.2 | 3 | 12 / 12 | No invasion |
| Mauritius (<1) | Kiwi | 0.05 | 3 | 12 / 12 | No invasion |
| Riccarton Bush (7) | Kiwi | 1.1 | 3 | 8 / 8 | Cat (**human error**) eradicated |
| Maungatautari north exclosure (35) | Kiwi | 2.8 | 3 | 11 / 11 | No invasion |
| Maungatautari south exclosure (65) | Kiwi | 3.5 | 3 | 11 / 11 | Rat (**tree fall**) eradicated |
| Tawharanui (660) | Kiwi | 2.8 | 3 | 11 / 8 | Mice, rat, stoat<br>**(Open fence ends at sea)**<br>Rats, stoat eradicated |
| Young Nicks Head (30) | Kiwi | 0.6 | 2 | 7 / 6 | Mice (**erosion damage**)<br>Mice re-established |
| Bushy Park (98) | Kiwi | 4.7 | 2 | 11 / 11 | Rat (**gate open**) eradicated |
| Godley Head (<1) | Kiwi | 0.05 | 2 | 2 / 2 | No invasion |
| Macrae's Flat (22) | Kiwi | 1.7 | 2 | 11 / 7 | Mice, stoat, possum<br>**(gate open, possible erosion)**<br>Stoat, possum eradicated<br>Mice re-established |
| Motu (<1) | Kiwi | 0.4 | 1 | 11/ 11 | No invasion |
| Horseshoe Bay (160) | Kiwi | 2.2 | 1 | 7 / 5 | Rats, White-tailed deer<br>**(Open fence ends at sea)** [1] |
| Mt Maungatautari (3,300) | Kiwi | 39 | 6 mon | 15 / 15 | Mouse (**tree fall**) eradicated |

[1] Fence ends are being modified in Spring 2007 to exclude rodent passage.

climbing abilities such as hedgehogs, ferrets, rabbits and hares. It did not effectively contain pest species with good climbing and jumping abilities. Others have also found the effectiveness of electric fences to be inconsistent for agile species such as possums (Clapperton and Matthews 1996, Day and Flight 2002), stoats (Day and MacGibbon 2002), cats (Long and Robley 2004) and fox (Poole and McKillop 2002). Electric fences are essentially psychological barriers that do not pose challenges beyond the physical ability of many species. As such, they can be crossed at will by any animal with sufficient motivation to so do, as demonstrated here by possums and cats receiving multiple shocks before escaping.

The mesh skirt at the base of the fence was highly effective at preventing animals from pushing or digging under the fence. Despite being more than physically capable of digging under, the animals did not perceive where the outer edge of the skirt began and preferentially focused digging attention at the base of the fence, on top of the skirt. Even after being housed in the experimental enclosures for up to three weeks, rabbits did not dig under the skirt. Our early data (unpublished), and that of others (e.g. Karori Wildlife Sanctuary, Inc. 1998) found that species such as rabbits could dig under up to 1 m of vertically buried mesh relatively easily. We also found that rats could dig under a mesh skirt if a log or similar object was placed on top of the skirt at its leading edge: they used the solid edge as a point to dig against and, once accidentally under the edge of the skirt, easily dug out. The use of a mesh skirt has become a standard feature of exclusion fences around the world after being proven to be the most successful method for rabbit fences in Australia over many years (Long and Robley 2004). However, mesh skirts do not always eliminate the problem of hole formation under a fence (e.g. Marks 1998, Fleming et al. 2001), so ongoing fence line maintenance is essential for continued fence integrity.

Exclusion of climbing and jumping animals in these experiments was achieved by the use of either a flat vertical sheet of steel, flexible plastic netting mounted on fibreglass rods (both on the Xcluder™ "Tui" design), or a smooth fixed steel hood placed at the top of the fence and protruding beyond the reach of the pest animals (Xcluder™ "Kiwi" fence). In all cases, these structures exceeded the physical abilities of the target pests. The 500 mm wide flat sheet of steel was too wide for mice, rats, stoats and possums to climb or jump across and when the top was placed at 1300 mm above the ground, it was too high for them to jump to the top. Similar physical limitations for these pest species have been observed by other researchers designing barrier fences (Karori Wildlife Sanctuary, Inc. 1998). Cats were extremely wary of climbing the unstable surface provided by the untensioned plastic netting (Day and MacGibbon 2002), preferring to let go rather than climb the netting once it began to collapse on them. Floppy wire mesh fences have been tried for cat exclusion in Australia (Coman and McCutchan 1994), but have had variable success, perhaps because of inconsistencies in mesh tensioning (Long and Robley 2004). When faced with the Xcluder™ Kiwi fence, cats jumped as high as 1800 mm directly from the ground. However, the hood at the top of the fence prevented animals from reaching or jumping to the top of the fence. A similar hood design was used successfully at Karori Wildlife Sanctuary to exclude pests (Karori Wildlife Sanctuary, Inc. 1998).

The behavioural patterns exhibited by pest animals trying to escape through, under or over fences in this research was similar to that observed by others. Most animals first attempted to escape by pushing through or under the fence (Lund and De Silva 1994, Long and Robley 2004). Therefore, the lower sections of the fence in particular must be meticulously constructed and maintained. The escape pressure on the fence was greatest at corners (especially inside angles), as animals walked or ran along the fence-line until they reached a corner and attempted to cross (Thompson 1979, Long and Robley 2004). Some animals in our trials and those of others appear to learn through trial and error to negotiate fences (Patterson 1977, Clapperton and Matthews 1996) and there was evidence of individuals learning to breach fences by watching successful breaches by conspecifics (Bird 1994, McKillop and Wilson 1999). In our research, Javan macaque that learned to cross substandard fence designs, subsequently assisted other members of the troop to cross the fence (Day 2004). Therefore, where learning and teaching are involved, the true effectiveness of a fence may not become apparent for a period of time after its construction. This has implications for the length of time over which experimental fence trials need to be conducted to ensure that animals that are initially deterred by a fence do not later learn to cross it (Long and Robley 2004). The in-situ efficacy data collected in this research provides good evidence that the fence designs tested were not overcome by animal learning.

Data from the in-situ fence sites clearly demonstrated two things. Firstly, the Xcluder™ "Tui" and Xcluder™ "Kiwi" fence designs are highly effective multispecies pest exclusion barriers. If constructed to exacting standards, the designs can be implemented in practice, withstand ongoing pressure from a suite of pest animals. Additional pest-proof components required to allow access while securing fence sites (e.g. double-doored pedestrian and vehicle gates, waterway gates for streams, etc.) were used at all in-situ sites and did not compromise the efficacy or integrity of the fenced areas unless accidentally left open.

Secondly, evidence from the in-situ sites highlighted that reinvasion of pests into areas protected by fences is a significant risk. All but two of the larger in-situ fence projects we have collected data from have had at least one invasion event since completion of their fence and eradication. Reinvasion risk events came from several sources, including human error, gates being left open, erosion damage and tree falls. The data suggests that reinvasion risk may be best considered as a matter of 'when', not 'if' (Day 2006), and proactive plans for managing reinvasion should be an integral part of any exclusion fencing project. These plans may include technology that minimises risk (e.g. the use of double-doored pedestrian and vehicle gates, gate alarms to alert managers to risk and remote surveillance systems to provide 'live' monitoring of fence integrity at all times) and staffing and infrastructure that enables immediate response. While most invasion events did not lead to re-establishment of pest populations, complete prevention of invasion would always be much better than curing an invasion problem after the fact.

When effective invasion risk management systems are implemented properly, reinvasion appears to be avoidable and fenced sanctuaries can be kept pest-free. Remote surveillance technology has been developed to immediately alert caretakers of risk events, such as tree falls, gates open, etc. At Mt. Maungatautari, multiple reinvasion risk events have occurred along the 47 km perimeter fence over several years, with none resulting in pest reinvasion if caretakers have responded to the risk immediately (within 3 hrs). Three pest invasion events were recorded at Maungatautari when immediate response plans were unable to be implemented for over 6 hrs. Ongoing research is being conducted at several exclusion fenced sites to further enhance the ability of projects to prevent invasion (see Speedy et al. 2007, this volume for details). This research includes measuring the risk of reinvasion by pest animals, understanding the behaviour of reinvaders, and designing the best methods for reinvasion prevention and (where required) cure.

The successful eradication of pest animals from within fenced areas (Speedy et al. 2007, this volume) and the fact that the areas have been kept pest-free for increasingly longer time periods has enabled exciting biological changes to begin. Research is underway on several fronts to measure these changes as they occur. One of the most significant early conservation gains from fence projects in New Zealand has been their use as pest-free havens for threatened species. Reintroduction of species such as North Island brown kiwi (*Apteryx australis mantelli*), black robin (*Petroica traversi*), takahe (*Notornis mantelli*) and tuatara (*Sphenodon punctatus*) to places from which they were long ago displaced by pest animals has occurred at several of the in-situ sites. The upsurge in exclusion fence projects in all major regions of New Zealand has led to significant and meaningful community engagement and education, with many projects being proposed, funded and driven by local communities, rather than by pest or conservation managers.

Further research with exclusion fencing is advancing on a number of fronts. The designs described here continue to be tested with additional pest species, such as fox and snakes. Surveillance technology continues to develop and fencing materials, methods and construction techniques are being improved continuously with experience. While it is still too early in the evolution of complete pest exclusion and eradication projects in New Zealand to fully quantify their true costs and benefits, the signs are encouraging. It appears that significant biological, social and economic gains are possible from these ambitious projects when appropriate exclusion technology, monitoring systems and expertise are used.

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## LITERATURE CITED

AVISS, M. AND A. ROBERTS. 1994. Pest fences: notes and comments. Department of Conservation, New Zealand, Threatened Species Occasional Publication No. 5.

BIRD, P. 1994. Improved electric fences and baiting techniques – a behavioural approach to integrated dingo control. Wood Research and Development Corporation, Project DAS 39.

BIRD, P., B. LOCK AND J. COOK. 1997. The Muloorina cell: a long-term trial to assess electric dingo fence designs. South Australian Dog Fence Board.

BURGETT, J., T. D. DAY, K. E. DAY, W. PITT AND R. SUGIHARA. 2007. From mice To mouflon: development and test of a complete mammalian pest barrier for Hawai'i. 15th Annual Hawaii Conservation Conference. Conservation strategies: matching science and management. July 2007, Honolulu, Hawaii, USA.

COMAN, B. J. AND J. MCCUTCHAN. 1994. Predator exclusion fencing for wildlife management in Australia. A report to the Australian Nature Conservation Agency.

CLAPPERTON, B. K. AND T. D. DAY. 2001. Cost-effectiveness of exclusion fencing for stoat and other pest control compared with conventional control. DoC Science Internal Series Department of Conservation, Wellington, New Zealand

CLAPPERTON, B. K. AND L. R. MATTHEWS. 1996. Trials of electric fencing for restricting the movements of common brushtail possums, *Trichosurus vulpecula* Kerr. Wildlife Research 23:571-579.

COWAN, P. E. AND D. S. RHODES. 1992. Restricting the movements of brushtail possums, *Trichosurus vulpecula*, on farmland with electric fencing. Wildlife Research 19:47-58.

DAY, T. D. 2004. Feasibility of pest proof fencing in Mauritius. Xcluder™ Pest Proof Fencing Company. Research Contract Report to the Mauritian Wildlife Foundation and Mauritian National Parks and Conservation Service.

DAY, T. D. 2006. Understanding and mitigating reinvasion risk at fenced sanctuaries. Proceedings of the Sanctuaries of New Zealand 2006 Annual Workshop, October, Kawhia, New Zealand.

DAY, T. D. AND K. E. FLIGHT. 2002. Evaluation of the efficacy of the North Cape fence as a possum barrier. Xcluder™ Pest Proof Fencing Company. Contract Report to the Department of Conservation.

DAY, T. D. AND R. J. MACGIBBON. 2002. Escape behaviour and physical abilities of vertebrate pests towards electrified and non-electrified fences. Xcluder™ Pest Proof Fencing Company unpublished report.

DUFTY, A. C., J. H. SEEBECK, J. MCKAY AND A. J. WATSON. 1994. Reintroduction of the eastern barred bandicoot, *Perameles gunnii* at Gellibrand Hill Park, Victoria. Pages 219–225 *in* M. Serena, editor. Reintroduction biology of Australian and New Zealand Fauna Surrey Beatty & Sons. New South Wales.

FLEMING, P., L. CORBETT, R. HARDEN AND P. THOMSON. 2001. Managing the impacts of dingoes and other wild dogs. Bureau of Rural Sciences, Canberra, Australia.

KARORI WILDLIFE SANCTUARY TRUST, INC. 1998. Karori Wildlife Sanctuary predator fence design: draft report on an investigation into, and development of an effective animal pest exclusion barrier. Karori Wildlife Sanctuary Trust Inc., Wellington, New Zealand.

KARORI WILDLIFE SANCTUARY TRUST, INC. 2001. Annual Report 2000-2001.

LONG, K. AND A. ROBLEY. 2004. Cost effective feral animal exclusion fencing for areas of high conservation value in Australia. Report for the Australian Government Department of the Environment and Heritage. Natural Heritage Trust.

LUND, R. AND S. DE SILVA. 1994. Animal behaviour needs consideration. Pages 65-67 *in* M. Casey, editor. Wires & pliers, the farm fencing manual. Kondinin Group, Belmont, WA, Australia.

MARKS, C. A. 1998. Field assessment of electric fencing to reduce fence damage by the common wombat *Vombatus ursinus*. Pages 298-304 *in* R. T. Wells and P. A. Pridmore, editors. Wombats. Surrey Beatty & Sons, Chipping Norton, United Kingdom.

MCKILLOP, I. G., S. G. V. GINELLA, C. J. WILSON, A. J. HANLON AND B. D. PUGH. 1993. The effects of power failure on the behaviour of European wild rabbits at electric fences. Applied Animal Behaviour Science 35:277-290.

MCKILLOP, I. G., AND R. M. SIBLEY. 1988. Animal behaviour at electric fences and the implications for management. Mammal Review 18:91-103.

MCKILLOP, I. G. AND C. J. WILSON. 1999. The behaviour of free-living European wild rabbits at electric fences. Crop Protection 18:193–197.

MOSEBY, K. E. AND E. O. O'DONNELL. 2003. Reintroduction of the greater bilby, *Macrotis lagotis* (Reid) (Marsupialia: Thylacomyidae), to northern South Australia: survival, ecology and notes on reintroduction protocols. Wildlife Research 30:15–27.

MINSKY, D. 1980. Preventing fox predation at a least tern colony with an electric fence. Journal of Field Ornithology 51:180-181.

PATTERSON, I. J. 1977. The control of fox movement by electric fencing. Biological Conservation 11:267–278.

POOLE, D. W. AND I. G. MCKILLOP. 2002. Effectiveness of two types of electric fence for excluding the Red fox (*Vulpes vulpes*). Mammal Review 32:51–57.

SEXTON, D. 1984. Electric fence policy. Working party report to the Minister for Conservation, Forests and Lands the Hon. R. A. MacKenzie, M.L.C., Victorian Department of Conservation, Forests and Lands.

SPEEDY, C., T. D. DAY AND J. INNES. 2007. Pest eradication technology – the critical partner to pest exclusion technology: the Maungatautari experience. Pages 115-126 *in* G. W. Witmer, W. C. Pitt, and K. A. Fagerstone, editors. Proceedings of the managing vertebrate invasive species. August 2007. Fort Collins, Colorado, USA.

TAYLOR R. H., G. W. KAISER AND M. C. DREVER. 2000. Eradication of Norway rats for recovery of seabird habitat on Langara Island, British Columbia. Restoration Ecology 8:151-160.

THOMPSON, B. C. 1979. Evaluation of wire fences for coyote control. Journal of Range Management 32:457–461.

TOWNS D. R. AND K. G. BROOMIE. 2003. From small maria to massive campbell: forty years of rat eradication from New Zealand islands. New Zealand Journal of Zoology 30:377-398.

VEITCH C. R. AND B. D. BELL. 1990. Eradication of introduced animals from the islands of New Zealand. *In* D. R. Towns, C. H. Daugherty and I. A. E. Atkinson, editors. Ecological restoration of New Zealand islands. Conservation Sciences Publication 2.

# WILDLIFE CONTRACEPTION: TARGETING THE OOCYTE

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***Abstract:*** The USDA's National Wildlife Research Center (NWRC) has successfully researched and developed a number of chemical and immunologically-based wildlife contraceptives. Diazacon™ interferes with cholesterol metabolism and disrupts steroidogenesis, while nicarbazin (registered as OvoControl-P® and OvoControl-G®) disrupts the structure of the vitelline membrane of bird eggs. Immunologically-based agents act to stimulate targeted antibody production. GonaCon™ causes the host's immune system to bind gonadotropin releasing hormone, preventing ovulation, while SpayVac™ prevents fertilization of the post-ovulatory oocyte. This kind of target specificity can be highly advantageous. A number of oocyte-only control contraceptive strategies are currently being researched at the NWRC. 4-Vinylcyclohexene diepoxide (VCD) is an industrial chemical which is specifically ovotoxic, depleting the ovarian oocyte pool with repeated exposure. Research into VCD efficacy as well as the comparability of a similar diepoxide, ERL 4221, in rats and pigs is in progress. Immunological inhibition of recently discovered oocyte-secreted proteins which regulate follicular development in mammals is also of interest. Two such proteins, growth differentiation factor 9 (GDF9) and bone morphogenic protein 15 (BMP15), are highly specific targets for the suppression or elimination of folliculogenesis. These oocyte-specific strategies may offer new, effective alternatives for wildlife contraception.

***Key Words:*** 4-vinylcyclohexene diepoxide, 20,25-diazacholesterol, Diazicon™, bone morphogenic protein 15, contraception, ERL 4221, GonaCon™, gonadotropin releasing hormone, growth diffentiation factor 9, invasive species, nicarbazin, oocyte, porcine zona pellucida, SpayVac™.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

For the last 15 years, the National Wildlife Research Center (NWRC) has been involved in the research and development of chemical and immunological contraceptive agents to control overabundant wildlife. Chemical agents inhibit or diminish reproduction by exerting an effect which is not mediated by the organism's immune system. These chemicals can be natural unmodified compounds, synthetic non-natural compounds, or biologically-produced molecules whose structures have been modified to alter their normal interactions in the body. Immunologically-based agents act by eliciting a response from the immune system in the form of antibody production directed at a reproductively important target such as a hormone. These molecular targets do not, by themselves, possess the immunogenicity necessary to stimulate the immune system, so the molecule or a small, immunologically active portion of it (an epitope) must be linked or conjugated to a much larger molecule that the immune system will detect, identify as foreign, and produce antibodies against. Ideally, antibodies to the target compound are produced as a by-product of this process. The selection of this larger carrier molecule can be as important as the identification of the appropriate epitope of the target molecule. Normally, the conjugate is dissolved in a saline solution and aggressively mixed with an adjuvant (a mixture of surfactant and mineral oil) to form a thick emulsion. When injected, the appropriate adjuvant potentiates the immune system, enhancing the response to the conjugate without causing serious localized tissue damage. The adjuvant also serves to immobilize and extend the release of the conjugate, maximizing its immunogenicity.

Recently, increasing attention at the NWRC as well as other institutions worldwide has been given to the application of contraceptive agents and strategies to control escalating numbers of introduced or invasive species. Additionally, the NWRC is currently researching new approaches to wildlife contraception utilizing both chemical and immunogenic agents whose actions focus on the pre-fertilization oocyte. Hopefully, the use of these

agents will compliment the chemical/immunogenic products already developed by the NWRC for the contraception of problem wildlife, particularly invasive species.

## EXISTING CONTRACEPTIVE STRATEGIES

### DiazaCon™

20,25 Diazacholesterol dihydrochloride (CAS# 1249-84-9) was originally developed as a cholesterol-lowering drug for use in humans (Sachs and Wolfman 1965) and later investigated to determine its efficacy in reducing the cholesterol content in egg yolks for human consumption (Singh et al. 1972). The chemical 20,25-diazacholesterol dihydrochloride (Figure 1) is an analog of cholesterol, in which carbons in the 20 and 25 positions have been replaced with nitrogen. It acts by mimicking cholesterol and inhibiting the $\Delta^{24}$-reductase enzyme which converts desmosterol to cholesterol during the process of steroidogenesis (Ahrens et al. 1965, Ranney and Cook 1965). This results in an increase in desmosterol and a corresponding decrease in cholesterol, which in turn reduces the concentrations of steroid sex hormones (e.g. testosterone and estrogen) that are critical for proper reproductive function. Used under a variety of synonyms and trade names such as SC-12937 and ornitrol, 20,25-diazacholesterol has been investigated as a contraceptive in several bird species (Johnston et al. 2001a). Under the new trade name DiazaCon™, the NWRC has continued extensive research using 20,25-diazacholesterol as a contraceptive in birds and mammals.

Yoder et al. (2005) summarized a series of laboratory experiments performed at the NWRC using DiazaCon™ as a contraceptive in quail *(Coturnix coturnix)*, ring-necked doves *(Streptopelia risoria)*, brown headed cowbirds

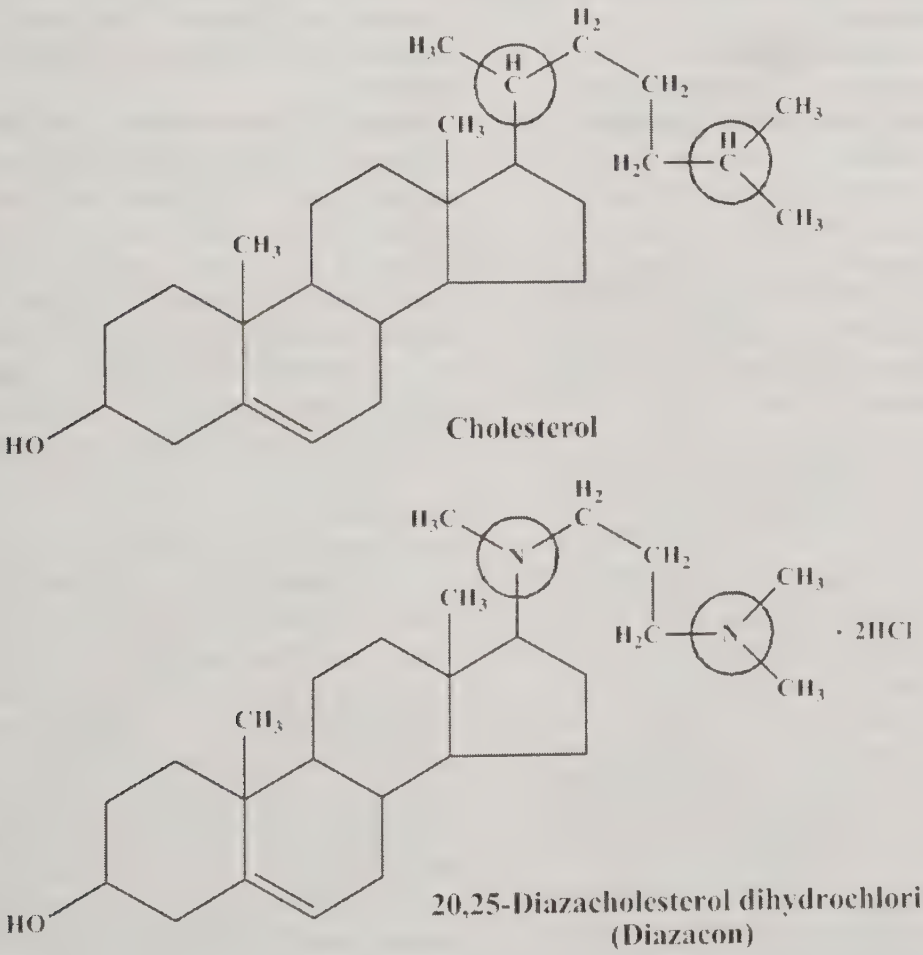



**Figure 1.** Structural comparison of cholesterol and 20,25-diazacholesterol (Diazacon™). Circled carbons in cholesterol are substituted with nitrogens in Diazacon™.

*(Molothrus ater)*, monk parakeets *(Myiopsitta monachus)*, American crows *(Corvus brachyrhyncus)*, and mallards *(Anas platyrhynchos)*. Serum levels of desmosterol and cholesterol as well as egg production (when possible) were assessed. In all species, treated birds showed substantially higher levels of serum desmosterol and reduced levels of cholesterol in treated birds compared to controls. The direct effect of DiazaCon™ treatment on egg production and hatchability could not be assessed in cowbirds or crows, but treatment dramatically decreased egg production and hatchability in monk parakeets, mallards, and quail. Treated quail also exhibited decreases in female serum progesterone and male testosterone levels of 42% and 37%, respectively (C. Yoder, unpublished data). Increased serum desmosterol levels with a concomitant decrease in serum cholesterol concentrations may serve as a useful indicator of decreased reproductive potential in situations where adults can be captured and blood samples taken, but nests are inaccessible or adults will not breed in captivity.

The use of 20,25-diazacholesterol as a mammalian contraceptive has also been explored. Spermatogenesis was inhibited in both bandicoot rats *(Bandicota bengalensis)* injected with 20,25-diazacholesterol at 100-200 mg $kg^{-1}$ (Hikim and Chakraborty 1986, Hikim 1987) and in mice *(Mus musculus)* given 28 doses of 10, 20, or 30 mg $kg^{-1}$ (Singh and Chakravarty 2003). Nash et al. (2007) found both a 47% decrease in reproductive success and significantly increased serum desmosterol and decreased serum cholesterol in individual free-ranging black-tailed prairie dogs *(Cynomys ludovicianus)* given daily portions of about 18 g of a treated oat baits containing about 45 mg DiazaCon™ for 10 days.

The contraceptive potential of DiazaCon™ is currently being investigated in a number of invasive species. Native to southern South America, the monk parakeet is now widely distributed globally. Between the late 1960s and early 1970s, more than 64,000 monk parakeets were imported into North America (Lever 1987). Parakeets were released both intentionally and unintentionally and stable populations currently exist in as many as 15 states in the United States (US, Spreyer and Bucher 1998). Monk parakeets have also become established throughout the Caribbean, Japan, Israel, and much of Europe. Although the parakeet may pose a threat to local agriculture, the primary problem associated with this species in the US is the construction of large communal nests, woven together out of twigs and branches. These nests are often built in electric utility substations and transmission lines and can cause short circuits and resultant power outages (Avery et al. 2002, Yoder et al. 2007). In the state of Florida, where the population of monk parakeets may exceed 100,000 individuals, the threat to power generation and transmission equipment is of serious concern. DiazaCon™ administered through gavage to captive monk parakeets elicited increased desmosterol serum levels with associated decreases in serum cholesterol as well as significantly decreased egg production in studies conducted by the NWRC (Yoder et al. 2007).

The rose-ringed parakeet *(Psittacula krameri)* is another invasive parrot which has been widely exported for the pet trade. The species native range covers much of northern and central Africa as well as the Indian sub-continent, but rose-ringed parakeets have established self-sustaining populations in the United Kingdom (UK), Japan, Iran, South Africa, and the US (primarily in California and Florida). Large flocks of parakeets can cause extensive damage to orchards and croplands.

The UK's Central Science Laboratory (CSL), in cooperation with the NWRC, is currently researching the use of DiazaCon™ for the control of this species. The CSL is also exploring DiazaCon™ as a means of controlling burgeoning populations of the eastern grey squirrel *(Sciurus carolinensis)*. Introduced in the late 18th century from the eastern US, the grey squirrel has rapidly expanded its range and has displaced the native Eurasian red squirrel *(Sciurus vulgaris)* throughout much of the UK.

**Nicarbazin**

Nicarbazin is an equimolar mixture of 4,4′-dinitrocarbanilide (DNC) and 2-hydroxy-6,6-dimethylpyrimidine (HDP, Figure 2). Commonly used as a treatment for coccidiosis in broiler chickens (Ott et al. 1956), nicarbazin consumption by laying or breeding hens causes a reduction in egg laying and hatchability (Jones et al. 1990, Hughes et al. 1991). Although the exact mechanism of nicarbazin action is unclear, it may prevent ova from maturing (Baker et al. 1957), probably by affecting the structural integrity of the vitelline membrane (the membrane which separates the egg's yolk from the albumen), allowing yolk and albumen to mix (Cunningham 1977, Chapman 1994). Decrease in egg production may be due to a lack of proper yolk deposition in the maturing follicle (Luck 1979). Yoder et al. (2006) found that

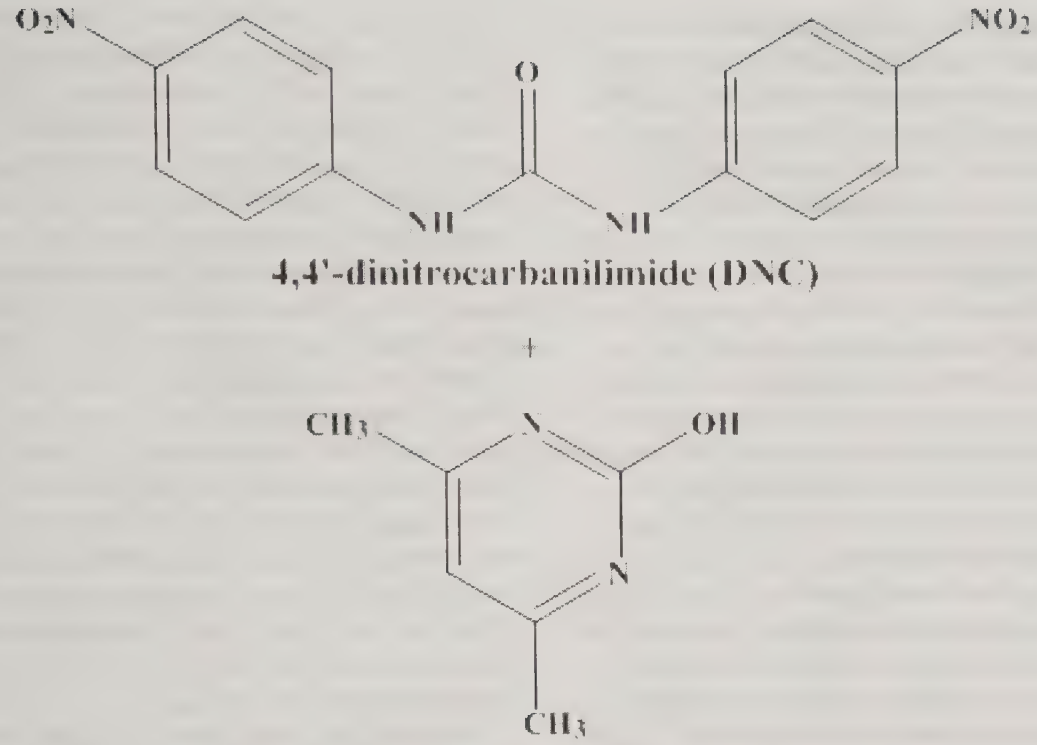



**Figure 2.** Chemical structure of nicarbazin, an equimolar mixture of 4,4'-dinitrocarbanilimide (DNC) and 2-hydroxy-6,6-dimethylpyrimidine (HDP).

nicarbazin increased lipoprotein lipase activity and acted as a calcium ionophore in vitro, suggesting possible mechanisms for the effects observed in vivo.

By decreasing the number of eggs layed and overall egg hatchability, nicarbazin's potential as an avian contraceptive has been extensively investigated. In tandem with the development of a palatable, nicarbazin-containing bait, analytical methods have been developed to confirm the presence of DNC in avian serum, egg yolk and albumin, and eggshells following ingestion of treated feed or baits (Johnston et al. 2001b, Primus et al. 2003, Stahl et al. 2003).

Sherwood et al. (1956) found a 10-50% decrease in egg fertility from leghorn hens fed nicarbazin in concentrations ranging from 6-700 mg $kg^{-1}$ of feed. Chickens fed varying levels of nicarbazin over 14 days showed positive correlations between amounts of nicarbazin consumed and residues in plasma and eggs (Johnston et al. 2001b), while reproductive success as measured by both hatchability and decreased reproduction rate decreased as amount of nicarbazin consumed increased. Yoder et al. (2006) demonstrated a positive dose response relationship between increasing levels of nicarbazin consumed and DNC residues in plasma, eggs, and feces of treated mallards.

A series of studies conducted at the NWRC led to the development of OvoControl G®, a 2500 ppm nicarbazin bait developed jointly by NWRC and Innolytics, LLC (Rancho Santa Fe, CA). When given to captive Canada geese *(Branta canadensis)*, OvoControl G® was both well-accepted and effective in producing plasma DNC levels in excess of those found to be minimally necessary to affect reproduction in geese (Bynum et al. 2005, 2007). In field studies, free feeding of OvoControl G® to 63 pairs of nesting Canada geese at 10 sites (5 control, 5 treated) in Oregon resulted in a 36% decrease in egg hatchability when compared with control sites (Bynum et al. 2007). OvoControl G® was subsequently registered with the US Environmental Protection Agency (EPA) for use prior to and during the breeding season to reduce hatching success in Canada geese.

In a preliminary study using captive rock pigeons *(Columba livia)* fed a variety of nicarbazin formulations, Avery et al. (2006) found that a 2,500 ppm nicarbazin bait did not significantly reduce egg hatchability, which may have been due to lower than expected plasma nicarbazin residues. A bait formulation of 5,000 ppm produced sufficiently high plasma levels to affect reproduction. In a subsequent study, rock pigeons fed 5,000 ppm baits did not exhibit reduced egg production, but egg viability was reduced by 59%, and productivity tended to be inversely correlated with plasma DNC concentrations. (Avery 2006). These results led to an EPA registration of the nicarbazin-containing bait, OvoControl P®, for use in controlling pigeon populations.

Nicarbazin has been investigated as a possible contraceptive for invasive populations of rose-ringed parakeets in the UK, but birds gavaged with nicarbazin at doses ranging from 8.4 to 18 mg $kg^{-1}$ showed virtually no plasma DNC residues. However, DNC analysis in fecal material yielded high concentrations, suggesting a lack of absorption from the gut. Comparative differences in psittacine digestive physiology may be responsible for the loss of nicarbazin efficacy (C. Yoder, personal communication).

**GonaCon™**

Gonadotropin releasing hormone (GnRH) is a decapeptide produced by the hypothalamus and transported in the bloodstream to the anterior pituitary where it stimulates specialized cells called gonadotrophs to produce luteinizing hormone (LH) and follicle stimulating hormone (FSH). These gonadotropic hormones are critical for the stimulation and precise timing of ovulation in female vertebrates and for testosterone production and spermatogenesis in males. Prevention of GnRH delivery to the pituitary gonadotrophs and subsequent lack of LH/FSH secretion can partially or completely disrupt these functions, leading to infertility.

By itself, GnRH is not immunogenic as it is both a natural, endogenous peptide which is not identified as foreign, and is simply too small to elicit attention from the immune system. Developed at the NWRC, GonaCon™ is a contraceptive vaccine comprised of the GnRH decapeptide conjugated to either keyhole limpet hemocyanin (KLH), a large protein extracted from the keyhole limpet *(Diodora cayenensis)*, or "blue protein" extracted from the Chilean abalone *(Concholepas concholepas)*. The immunogenicity of the conjugate is increased by emulsifying it in AdjuVac™, an adjuvant also developed at the NWRC. AdjuVac™ is a modification of the Johne's vaccine made more immunogenic by the addition of killed *Mycobacterium avium*, a common bacteria (Miller et al. 2004). In rabbits, AdjuVac™ yielded a good immune response with fewer inflammatory reactions and associated tissue damage when compared with Freund's Complete, a frequently used adjuvant (Powers et al. 2007)

GonaCon™ is an effective contraceptive in a variety of species, including free-ranging California ground squirrels (*Spermophilus beecheyi*), captive Norway rats (*Rattus norvegicus*), domestic cats (*Felis catus*), domestic and feral swine (*Sus scrofa*), wild horses (*Equus caballus*), bison (*Bison bison*), and white-tailed deer *(Odocoileus virginianus)* [Miller et al. In Press A]. EPA registration will be pursued for contracepting female white-tailed deer (Fagerstone et al. In Press).

Feral pigs (*Sus scrofa*) have become invasive in 38 states, with an estimated 2 million individuals in Texas alone and have established wild populations in the UK, Australia, New Zealand, and South America. GonaCon™ was injected into captive domestic pigs in one of three vaccination injection treatments, either 800 µg, 1,600 µg, or a prime injection (400 µg) followed by a booster injection (400 µg). In all pigs, serum antibody titers were highest in the 2 x 400 µg injection treatment, followed by the 1,600 µg and 800 µg treatments. Antibody titers were inversely correlated with fertility and estrus suppression in sows and with testicular size and serum testosterone in boars (Miller et al. 2003). Killian et al. (2003) gave a single injection of either 1,000 µg or 2,000 µg GonaCon™ to penned feral pigs, and found reduced testicular and ovarian size as well as decreased serum testosterone and progesterone, all of which correlated with increasing antibody titers. Pregnancy was reduced by 90% using the 2,000 µg dosage, while 1,000 µg was sufficient to maximize effects in boars. GonaCon™ is currently being used in a research program to contracept feral pigs in the UK (Massei et al. In Press).

Brushtailed possums *(Trichosurus vulpecula)* are native to Australia and have been introduced to both the UK and New Zealand. In New Zealand, possums spread disease and consume native plants, bird eggs, and chicks. Researchers are currently employing GonaCon™ to determine its effectiveness in contracepting possums (Eckery et al. In Press).

**SpayVac™**

The zona pellucida (ZP) is a glycoprotein coat that encapsulates the ovum, and is essential for oocyte survival and the process of fertilization. In most mammals, the ZP consists of three large glycoproteins, but for the purposes of immunocontraception the principle ZP glycoprotein constituent is ZP3. This glycoprotein contains carbohydrate groups that function as sperm receptors, adhering to specific proteins on the surface of the sperm acrosomal membrane, anchoring the sperm to the ZP surface, and initiating the process of fertilization.

Porcine zone pellucida (PZP) is an extracted, purified preparation from pig ovaries. Injection of PZP into non-porcine species causes an immunogenic reaction, stimulating the production of antibodies directed against the zona's glycoproteins. This in turn prevents sperm from binding to the zona surface, resulting in infertility. Additionally, because the amino acid sequences of the zona glycoproteins are highly conserved between mammalian species, PZP-produced antibodies will frequently cross-react with the ZP of many other species (Killian et al. 2004). Several PZP preparations have been studied, but the use of SpayVac™ (Brown et al. 1997) emulsified in AdjuVac™ adjuvant yielded effective contraception in white-tailed deer for a period of 6 years with a single vaccination (Miller et al. In Press B). These results indicate that the preparation methodology used to produce SpayVac™ provides a "self-boosting" quality to the immunization.

Native species-specific ZP, non-native ZP, and PZP have been used to contracept a wide variety of species. Rats treated with mouse zona pellucida exhibited a decrease in both numbers of litters and pups per litter (Miller et al. 1997). Kirkpatrick et al. (1996) evaluated PZP in 74 species of captive zoo mammals and achieved successful contraception in 27 species from 8 families, including ursids, felids, mustelids, giraffids and bovids. The list of species in which separate studies have shown PZP to provide effective contraception is extensive and includes African elephants *(Loxodontia africana*, Fayer-Hosken et al. 1999), grey seals *(Halichoerus grypus,* Brown et. al 1997), coyotes (*Canis latrans*, Miller et al. 2006), burros (*Equus asinus*, Liu et al. 1989), and baboons (Dunbar et al. 1989). PZP does not appear to be effective in domestic cats (Gorman et al. 2002) which may be due to large differences in zona proteins between the donor and recipient species. Currently, SpayVac™ is used to control overabundant species such as white-tailed deer (Locke et al. 2007, Miller et al. In Press B) and wild horses (Killian et al. 2004), but no applications involving invasive species have been reported.

One of the advantages of the immune response to PZP is the targeting of specific zona glycoproteins which do not occur elsewhere in the body. The PZP response is also inherently sex-specific, as males to do not produce ZP proteins. Likewise, nicarbazin only affects female birds. Although other agents described in this paper may have specific action sites, they can produce more generalized, secondary effects which involve non-target tissues or physiological systems, possibly leading to undesirable side effects. Sex-specificity can be readily achieved for all contraceptive techniques if the desired individuals can be trapped and treated or vaccinated, it would be far more difficult if contraceptive agents were broadcast generally in the environment through baiting. In many, if not most, applications involving wildlife contraception, appropriate baits and baiting techniques will be essential for the efficient deployment of any contraceptive agent. Chemical agents such as DiazaCon™ or nicarbazin are usually better suited to inclusion within a bait matrix and are routinely presented in that form, but for immunologically-based agents, the preservation of immunogenicity following consumption and digestion poses significant developmental challenges. Conjugated agents such as GonaCon™ may prove more amenable to this approach than heterogeneous preparations like SpayVac™. Techniques that facilitate the effective oral delivery of contraceptives to free-ranging wildlife are also under development at the NWRC.

Difficulties related to target specificity might be reduced or eliminated with the continued development of contraceptive agents with highly specific action sites. The advantages of increased target specificity include: (1) selection of the sex of the individual to be contracepted, (2) location of action within the body, and (3) timing of the intervention. A variety of new approaches which specifically target the oocyte are now being researched at the NWRC.

## OOCTYTE-SPECIFIC STRATEGIES

**4-vinylcyclohexene diepoxide, ERL-4221**

As with the previous contraceptive agents discussed, the avenues of research being pursued by the NWRC include both chemical and immunological agents. 4-vinylcyclohexene (VCH)

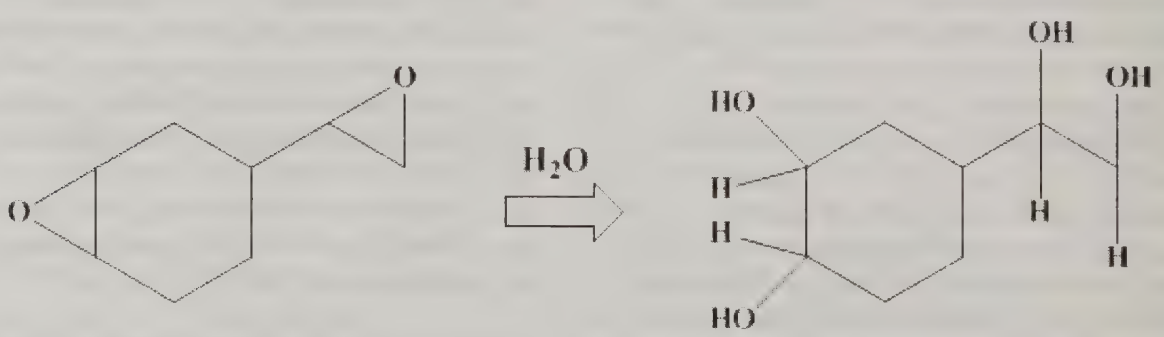
is a byproduct in the synthesis of rubber, insecticides and plasticizers. In vivo, VCH is metabolized to 4-vinylcyclohexene diepoxide (VCD, Smith et al. 1990a). Initial experiments showed that the administration of VCH to mice reduced the number of primary and secondary ovarian follicles (Collins and Manus 1987), but this effect was later found to be due to VCD following metabolization of VCH (Smith et al. 1990b). VCD ootoxicity is thought to be due to the reduced ability of primary and pre-antral follicles to convert VCD to its tetrol metabolite (Figure 3), resulting in follicular atresia by apoptosis (Hoyer et al. 2001). Intraperitoneal injection of VCD daily for 6 to 15 days at dosages of 80-240 mg $kg^{-1}$ results in significant or near-complete depletion of the ovarian oocyte pool in mice and rats (Hoyer et al. 2001), a phenomenon which has been used to model menopause in humans. VCD-induced oocyte depletion has also been observed in cynomolgous macaques (Appt et al. 2006) and dogs (Miers et al. 2005).

Because of various safety concerns associated with VCD usage, bulk production of the compound was terminated in 2005 (Ellis 2006). Available now only as a specialty chemical, the cost of VCD in large quantities has become prohibitive. For commercial uses, however, VCD was replaced by Cycloaliphatic Epoxide Resin, ERL-4221, Dow Chemical Co. (Figure 4). ERL-4221 is significantly less hazardous, but there are no reports on ERL-4221 efficacy as an ootoxicant. The potential of ERL-4221 as an oocyte-targeted contraceptive tool is currently being investigated by the NWRC in both feral pigs in cooperation with Texas A&M University, Kingsville, Texas, and in comparison with VCD in rats at NWRC headquarters in Fort Collins, Colorado.

**Growth Differentiation Factor 9, Bone Morphogenic Protein 15**

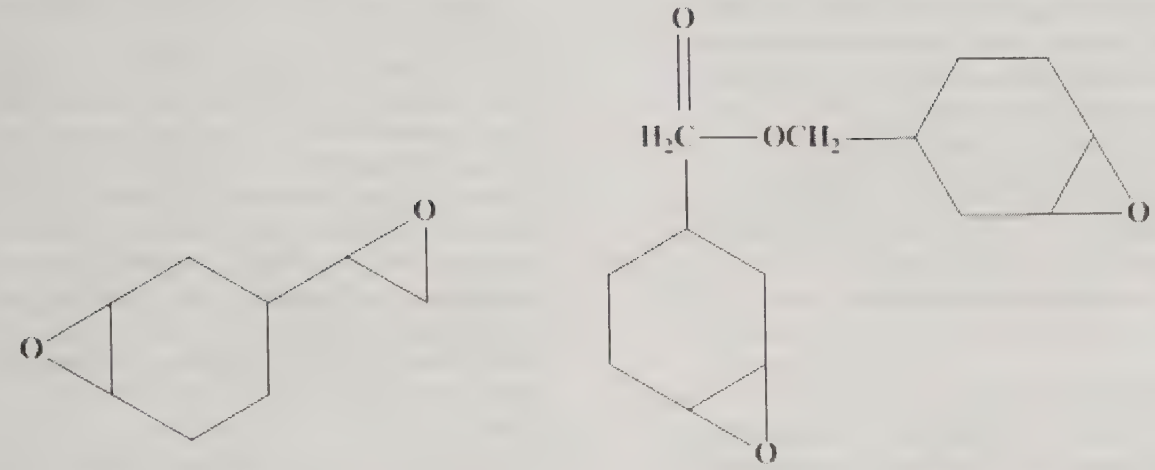
The control of fertility in general and follicular development in particular is an extremely complex process that is not well understood. A wide variety of biochemical signals mediate communication between the developing oocyte and follicle and the ovarian granulosa cells which surround and support the growing follicle. One group of large, structurally related proteins which have been shown to be extensively involved in this process is the transforming growth factor, β superfamily. The superfamily can be broken down into a number of subgroups, among which are the growth determination factors (GDF) and the bone morphogenic proteins (BMP). These proteins are macromolecules, with some as large as 450+ amino acid units (mers). Some twenty BMP (both cytokines and growth factors) have been identified and are thought to play important roles in embryonic patterning and early skeletal development, while some fifteen GDF have been characterized and have regulatory functions in the oocyte and virtually all developing and differentiating embryonic tissues.

During folliculogenesis, several primordial follicles, each containing a primary oocyte, are recruited for development. The oocyte increases in size and is encapsulated by the glycoprotein zona pellucida coat, and the surrounding granulosa cells




**4-vinylcyclohexene diepoxide (VCD)** **VCD tetrol**

**Figure 3.** Chemical structure of 4-vinylcyclohexene diepoxide and its tetrol metabolite.




**4-vinylcyclohexene diepoxide (VCD)** **Cycloaliphatic Epoxide Resin (ERL-4221)**

**Figure 4.** Structural comparison of 4-vinylcyclohexene diepoxide (VCD) and Cycloaliphatic Epoxide Resin ERL-4221.

proliferate, change shape, and begin secreting fluid, marking the formation of the primary follicle. Follicular cells continue to differentiate, while fluid pockets begin to form between the granulosa cells, marking the pre-antral stage of follicular development. These fluid pockets merge, forming a large, liquid filled antral or secondary follicle. The oocyte has become many times its original size, and the granulosa cells which now line the antral cavity have also differentiated into the cumulus oophorous, which attaches the oocyte to the antral wall, and the corona radiata, comprised of support cells which surround the oocyte. Of the follicles which began this process, all but one have died and become atretic. The remaining follicle becomes pre-ovulatory in preparation for ovulation, followed by the resumption of meiosis in the oocyte and the possibility of fertilization.

Oocyte-produced growth factors play a critical role in folliculogenesis, mediating communication between the oocyte and granulosa/granulosa-derived cells. Two of these factors, growth differentiation factor 9 (GDF9) and bone morphogenic protein 15 (BMP15), have been identified as particularly important. They interact to regulate proliferation and differentiation of granulosa cells, probably via production of inhibin and interaction with FSH. In most mammalian species, GDF9 appears to be the dominant factor as follicular development is completely halted in its absence, but both factors are usually necessary for successful folliculogenesis, ovulation, and resumption of meiosis.

Unlike small peptides like GnRH, successful development of a contraceptive vaccine using large macroproteins is made more difficult and expensive because of the sheer size of the molecule and the problems involved in its synthesis. Frequently, however, an epitope of the macromolecule is recognized by the immune system, and can be synthesized and conjugated to a larger antigenic molecule. Using the known amino acid sequences of both GDF9 and BMP15 in sheep, McNatty et al. (2006) synthesized 9 (for BMP15) or 10 (for GDF9) sequential, overlapping 9-15 mer epitopes starting from close to the carboxylic acid-containing (c-terminal) end and progressing approximately 100 mer towards the amine (n)-terminal end of each molecule to compare immunogenicity and contraceptive effectiveness of each epitope. As with GnRH, these epitopes were too small to be immunogenic, so they were conjugated to KLH. Using these conjugates, sheep ewes were immunized with either anti-GDF9 or anti-BMP15 and the epitopes exerting the greatest contraceptive effectiveness were identified. Overall, McNatty et al. (2006) concluded that anti-GDF9 and anti-BMP15 showed promise as contraceptive agents. In cooperation with New Zealand's Victoria University and White Buffalo, Inc. (Hamden, CT), a non-profit wildlife management organization specializing in non-lethal

control of white-tailed deer, the NWRC has obtained and conjugated both the effective GDF9 and BMP15 epitopes described by McNatty et al. (2006) and is currently exploring the use of these epitopes as contraceptive techniques.

In conjunction with the wide variety of contraceptive agents already developed by the NWRC, these new oocyte-specific contraceptive strategies may provide additional tools for the control of invasive species and other problem wildlife.

## LITERATURE CITED


AHRENS, R. A., J. DUPORT, AND M. J. THOMPSON. 1965. Sterols in brain and liver of young rats fed 20,25-diazacholesterol. Proceedings of the Society for Experimental Biology and Medicine 118:436-438.

APPT, S. E., J. R. KAPLAN, T. B. CLARKSON, J. M. CLINE, P. J. CHRISTIAN, AND P. B. HOYER. 2006. Destruction of primordial ovarian follicles in adult cynomolgus macaques after exposure to 4-vinlycyclohexene diepoxide: a nonhuman primate model of the menopausal transition. Fertility and Sterility 86 (Supplement 1):1210-1216.

AVERY, M. L., E. C. GREINER, J. R. LINDSAY, J. R. NEWMAN, AND S. PRUETT-JONES. 2002. Monk parakeet management at electric utility facilities in south Florida. Proceedings of the Vertebrate Pest Conference 10:140-145.

AVERY, M. L., K. L. KEACHER, AND E. A. TILLMAN. 2006. Development of nicarbazin bait for managing rock pigeon populations. Proc. 22nd Vertebrate Pest Conf. (R.M. Timm and J.M. O'Brien, Eds.) published at Univ. of Calif., Davis. Pp. 116-120.

AVERY, M. L. 2006. Effects of nicarbazin bait on pigeon reproduction. Unpublished report: QA-1329. National Wildlife Research Center, Gainesville, Florida, USA.

BAKER, R. C., R. W. HILL, A. VAN TIENHOVEN, AND J. H. BRUCKNER. 1957. The effect of nicarbazin on egg production and egg quality. Poultry Science 36:718-726.

BROWN, R. G., W. D. EDDINGTON, J. D. EDDINGTON, W. C. KIMMINS, M. MEZEI, J. L. PARSON, AND B. POHAJDAK. 1997. Evidence for a long-lasting single administration vaccine in wild grey seals. Journal of Reproductive Immunology 35:43-51.

BYNUM, K. S., C. A.YODER, J. D. EISEMANN, J. J. JOHNSTON, AND L. A. MILLER. 2005. Development of nicarbazin as a reproductive inhibitor for resident Canada geese. Pages 179-189 *in* D. L. Notle and K. A. Fagerstone, editors. Proceedings of the 11th Wildlife Damage Management Conference.

BYNUM, K. S., J. D. EISENMANN, G. C. WEAVER, C. A. YODER, K. A. FAGERSTONE, AND L. A. MILLER. 2007. Nicarbazin OvoControl G bait reduces hatchability of eggs laid by resident Canada geese in Oregon. Journal of Wildlife Management 71:135-143

COLLINS, J. J. AND A. G. MANUS. 1987. Toxicological evaluation of 4-vinylcyclohexene. I. Prechronic (14-day) and subchronic (13-week) gavage studies in Fischer 344 rats and $B6C3F_1$ mice. Journal of Toxicology and Environmental Health 21:493-505.

CHAPMAN, D. H. 1994. A review of the biological activity of the anticoccidial drug nicarbazin and its application for the control of coccidiosis in poultry. Poultry Science Review 5:231-243.

CUNNINGHAM, F. E. 1977. Properties of albumen from eggs having mottled yolks. Poultry Science 56:1819-1821.

DUNBAR, B. S., C. LO, J. POWELL, AND V. C. STEVENS. 1989. Use of a synthetic peptide adjuvant for the immunization of baboons with denatured and deglycosylated pig zona pellucida glycoproteins. Fertility and Sterility 52:311-38.

ECKERY, D. C., B. P. THOMPSON, AND L. A. MILLER. In Press. Effects of immunisation against GnRH on fertility of the brushtail possum. Wildlife Research.

ELLIS, E. A. 2006. Solutions to the problem of substitution of ERL-4221 for vinyl cyclohexene dioxide in Spurr low viscosity embedding formulations. Microscopy Today 14:32-33.

FAGERSTONE, K. A., L. A. MILLER, J. D. EISEMANN, AND J. P. GIONFRIDDO. In Press. Development and registration of GonaCon™ immunocontraceptive vaccine in the United States. Wildlife Research.

FAYER-HOSKIN, R. A., H. J. BERTSHINGER, J. F. KIRKPATRICK, D. GROBLER, N. LAMBERSKI, G. HONNEWMAN, AND T. ULRICH. 1999. Contraceptive potential of the porcine zona pellucida vaccine in the African elephant *(Loxodontia africana)*. Theriogenology 52:835:846.

GORMAN, S. P., J. K. LEVY, A. L. HAMPTON, W. R. COLLANTE, A. L. HARRIS, AND R. G. BROWN. 2002. Evaluation of a porcine zona pellucida vaccine for the immunocontraception of domestic kittens (*Felis catus*). Theriogenology 58:135:149.

HIKIM, A. P. S. AND J. CHAKRABORTY. 1986. Effects of diazacholesterol dihydrochloride (SC-12937), an avian antifertility agent, on rat testis. Journal of Andrology 7:277-284.

HIKIM, A. P. 1987. Effect of 20,25-diazacholesterol dihydrochloride on the spermatogenesis of a wild rat, Bandicota bengalensis. International Journal of Fertilility 32:320-323.

HOYER, P. B., P. DEVINE, X. HU, K. E. THOMPSON, AND I. G. SIPES. 2001. Ovarian toxicity of 4-vinylcyclohexene diepoxide: A mechanistic model. Toxicologic Pathology 29:91-99.

HUGHES, B. L., J. E. JONES, J. E. TOLER, J. SOLIS, AND D. J. CASTALDO. 1991. Effects of exposing broiler feeders to nicarbazin contaminated feed. Poultry Science 70:476-482.

Johnston, J. J., M. J. Goodall, J. C. Hurley, C. A. Yoder, and L. A. Miller. 2001a. Determination of DiazaCon in quail feed and quail serum by ion pair reversed-phase chromatography. Journal of the Association of Official Analytical Chemists International 84:634-639.

Johnston, J. J., W. M. Britton, A. MacDonald, T. M. Primus, M. J. Goodall, C. A. Yoder, L. A. Miller, and K. A. Fagerstone. 2001b. Quantification of plasma and egg 4,4′-dinitrocarbanilide (DNC) residues for the efficient development of a nicarbazin-based contraceptive for pest waterfowl. Pest Management Science 58:197-202.

Jones, J. E. J. Solis, L. Hughes, D. J. Castaldo, and J. E. Toler. 1990. Reproduction responses of broiler-breeders to anticoccidial agents. Poultry Science 69:27-36.

Killian, G., L. Miller, J. Rhyan, T. Dees, D. Perry, and H. Doten. 2003. Evaluation of GnRH contraceptive vaccine in captive feral swine in Florida. *In* K. A. Fagerstone and G. W. Witmer, editors. Proceedings of the 10th Wildlife Damage Management Conference.

Killian, G., L. A. Miller, N. K. Diehl, J. Rhyan, and D. Thain. 2004. Evaluation of three contraceptive approaches for population control of wild horses. Pages 263-268 *in* R. M. Timm and W. P. Gorenzel, editors. Proceedings of the 21st Vertebrate Pest Conference. University of California, Davis, California, USA.

Kirkpatrick, J. F., J. W. Turner Jr., I. K. Liu, and R. Fayer-Hosken. 1996. Applications of pig zona pellucida immunocontraception to wildlife fertility control. Journal of Reproduction and Fertility Supplement 50:183:189.

Lever, C. 1987. Naturalized birds of the world. Longman Scientific & Technical, London, United Kingdom.

Locke, S. L., M. W. Cook, L. A. Halveson, D. S. Davis, R. R. Lopez, N. J. Silvy, and M. A. Fraker. 2007. Effectiveness of Spayvac® for reducing white-tailed deer fertility. Journal of Wildlife Diseases 43:726-730.

Liu, I. K. M., M. Bernoco, and M. Feldman. 1989. Contraception in mares hetero-immunized with porcine zona pellucida. Journal of Reproduction and Fertility 85:19-29.

Luck, M. R. 1979. The adverse effects of nicarbazin on reproductive activity in the hen. British Poultry Science 20:605-607.

Massei, G., D. P. Cowan, and L. A. Miller. In Press. Fertility control to manage wild boar using the GnRH vaccine GonaCon™. Wildlife Research.

McNatty, K. P., N. L. Hudson, L. Whiting, K. L. Reader, S. Lun, A. Western, D. A. Heath, P. Smith, L. G. Moore, and J. L. Juengel. 2006. The effects of immunizing sheep with different BMP15 or GDF9 peptide sequences on ovarian follicular activity and ovulation rate. Biology of Reproduction 76:552-560.

Miers, H., C. Dyer, R. Craig, D. Kalleco, K. Karr, Z. Robinson, S. Marion, P. Hoyer, C. Blalock, and L. Mayer. 2005. 4-Vinylcyclohexene diepoxide-induced primordial follicle depletion in canine ovaries. Abstract presented at The Society of the Study of Reproduction, 38th Annual Meeting, July 24 – 27, 2005, Quebec City, Quebec, Canada

Miller, L. A., B. E. Johns, D. J. Elias, and K. A. Crane. 1997. Comparative efficacy of two immunocontraceptive vaccines. Vaccine 15:1858-1862.

Miller, L., J. Rhyan, and G. Killian. 2003. Evaluation of GnRH contraceptive vaccine using domestic swine as a model for feral hogs. Pages 120-127 *in* K. A. Fagerstone and G. W. Witmer, editors. Proceedings of the 10th Wildlife Damage Management Conference.

Miller, L. A., J. Rhyan, and G. Killian. 2004. GonaCon™, a versatile GnRH contraceptive for a large variety of pest animal problems. Pages 269-273 *in* R. M. Timm and W. P. Gorenzel, editors. Proceedings of the 21st Vertebrate Pest Conference. University of California, Davis, California, USA.

Miller, L. A., K. Bynum, and D. Zemlicka. 2006. PZP immunocontraception in coyotes: a multi-year study with three vaccine formulations. Pages 88-95 *in* R. M. Timm and J. M. O'Brien, editors. Proceedings of the 22nd Vertebrate Pest Conference. University of California, Davis, California, USA.

Miller, L. A., J. P. Gionfriddo, K. A. Fagerstone, J. C. Rhyan, and G. J. Killian. In Press A. The single-shot GnRH immunocontraceptive vaccine (GonaCon™) in white-tailed deer: comparison of several GnRH preparations. American Journal of Reproductive Immunology.

Miller. L. A. K. A. Fagerstone, K. S. Bynum, and G. J. Killian. In Press B. Factors contributing to the success of a single-shot, multi-year PZP immunocontraceptive vaccine for white-tailed deer. Vaccine.

Nash, P., C. A. Furcolow, K. S. Bynum, C. A. Yoder, L. A. Miller, and J. J. Johnston. 2007. 20, 25-Diazacholesterol as an oral contraceptive for black-tailed prairie dog population management. Human-Wildlife Conflicts 1:60-67.

Ott, W. H., S. Kuna, C. C. Porter, and A. C. Cuckler. 1956. Biological studies on nicarbazin, a new anticoccidial agent. Poultry Science 35:1355-1367.

Powers, J. G., P. B. Nash, J. C. Rhyan, C. A. Yoder, and L. A. Miller. 2007. Comparison of immune and adverse effects induced by Adjuvac and Freund's complete adjuvant in New Zealand white rabbits *(Oryctolagus cuniculus)*. Lab Animal 36:51-58.

PRIMUS, T. M., D. J. KOHLER, M. A. GOODALL, C. A.YODER, T. MATHIES, L. MILLER, J. J. JOHNSTON, AND K. VERCAUTEREN. 2003. Chromatographic determination of 4,4'-Dinitrocarbanilide, the active component of the infertility agent nicarbazin, in chicken, duck, goose and snake eggs Journal of Association of Official Analytical Chemists International 86:1144-1148.

RANNEY, R. E. AND D. I. COOK. 1965. The hypocholesterolemic action of 20,25-diazacholesterol. Archives Internationales de Pharmacodynamie et de Therapie 154:51-62.

SACHS, B. A. AND L.WOLFMAN. 1965. 20, 25-Diazacholesteroldihydrochloride inhibition of cholesterol biosynthesis in hyperlipemic subjects. Archives of Internal Medicine 116:366-372.

SHERWOOD, D. H., T. T. MILBY, AND W. A. HIGGINS. 1956. The effect of nicarbazin on reproduction in white rock breeder hens. Poultry Science 35:1014-1019.

SINGH, R. A., J. F. WEISS, AND E. C. NABER. 1972. Effect of azasterols on sterol metabolism of the laying hen. Poultry Science 51:449-457.

SINGH, S. K., AND S. CHAKRAVARTY. 2003. Antispermatogenic and antifertility effects of 20,25-diazacholesterol dihydrochloride in mice. Reproductive Toxicology 17:37-44.

SMITH, B. J., D. E. CARTER, AND I. G. SIPES. 1990a. Comparison of the disposition and in vitro metabolism of 4-vinylcyclohexene in the female mouse and rat. Toxicology and Applied Pharmacology 105:364-371.

SMITH, B. J., D. R. MATTISON, AND I. G. SIPES. 1990b. The role of epoxidation in 4-vinylcyclohexene-induced ovarian toxicity. Toxicology and Applied Pharmacology 105:372-381.

SPREYER, M. F. AND E. H. BUCHER. 1998. Monk parakeet (*Myiopsitta monachus*). Birds of North America 322:1-23.

STAHL, R. S., K. C. VERCAUTEREN, T. L. BUETTGENBACH, AND J. J. JOHNSTON. 2003. Determination of 4,4'-dintrocarbanilide (DNC), a component of nicarbazin, in Canada goose *(Branta canadensis)* eggshells using high-performance liquid chromatography. Journal of Agriculture and Food Chemistry 51:1130-1135.

YODER, C. A., K. S. BYNUM, AND L. A. MILLER. 2005. Development of DiazaCon™ as an avian contraceptive. Proceedings of the 11th Wildlife Damage Management Conference 11:190-201.

YODER, C. A., J. K. GRAHAM, L. A. MILLER, K. S. BYNUM, J. J. JOHNSTON, AND M. J. GOODALL. 2006. Evaluation of nicarbazin as a potential waterfowl contraceptive using mallards as a model. Poultry Science 85:1275-1284.

YODER, C. A., M. L. AVERY, K. L. KEACHER, AND E. A. TILLMAN. 2007. Use of DiazaCon™ as a reproductive inhibitor for monk parakeets *(Myiopsitta monachus)*. Wildlife Research 34:80-83.

# METHODS AND STRATEGIES FOR MANAGING FERAL HOG DAMAGE IN GRAIN PRODUCTION AREAS IN CENTRAL TEXAS

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***Abstract:*** Texas Wildlife Services (WS) personnel conduct feral hog damage management projects to protect a variety of resources in Texas. For the past 12 years, WS personnel have conducted feral hog (*Sus scrofa*) damage management projects on an 11,000 acre wildlife management area (WMA) and surrounding farmlands located in central Texas. The WMA is managed by the United States (US) Army Corp of Engineers and Texas Parks and Wildlife Department. Surrounding areas are farmlands and the primary crops grown there are corn and grain sorghum. Feral hogs have damaged area grain crops and the US Army Corp of Engineers property. Annual projects have been conducted to reduce feral hog damage to crops and property. WS personnel have used an assortment of methods and strategies in a management plan to reduce feral hog damage in the area. This paper is a review of those methods, strategies, and the plan.

***Key Words:*** feral hog, invasive species, IPM, strategy, *Sus scrofa,* wildlife.

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## INTRODUCTION

Feral hog (*Sus scrofa*) management has received much attention in recent years as hog populations have increased and their distribution has expanded across Texas. Various strategies have been discussed, suggested, and reviewed in the literature, however, few have been implemented and described. Such a strategy has been in place at a site in central Texas since 1995 under the direction of the United States (US) Department of Agriculture's (USDA) Texas Wildlife Services (WS). The basic management strategy was developed in 1995 and it continues to evolve in an effort to maximize its effectiveness. Initial phases involved developing an integrated pest management (IPM) plan and obtaining cooperation from concerned parties. Littauer (1993) recommended an integrated approach to managing feral hog damage. Factors considered when developing the management plan were (1) seasonal timing of feral hog damage, (2) size of the management area, (3) number of WS personnel and amount of equipment that could be assigned to the specific project, and (4) annual variation in the local hog population density due to weather and area crop production. As the management plan has progressed, WS field personnel have identified other factors related to feral hog management in grain production areas. These factors have influenced the development of the management plan and include problems associated with detection of feral hog damage and conducting hog management activities in grain fields.

## MANAGEMENT AREA

The area includes approximately 22,000 ac (8,903 ha) of black land prairie and associated bottomlands in Williamson County, Texas. Approximately 11,000 ac (4,452 ha) in the area are US Army Corps of Engineers property, which are managed by Texas Parks and Wildlife Department (TPWD) as a wildlife management area (WMA). The US Corps of Engineers property includes Granger Lake, a 4,400 ac (1,780 ha) flood control reservoir fed by the San Gabriel River. The remaining lands (approximately 11,000 ac or 4,452 ha) are privately owned lands. Much of the private land is managed for agricultural production, including corn, grain sorghum, cotton, and livestock. Rainfall averages 34 in (86.36 cm) annually and ranged from 19.51 in (49.53 cm) to 51.29 in (130.28 cm) during the 12-year period.

## HISTORY OF FERAL HOG DAMAGE

Farmers in the area have experienced feral hog damage to crops and pasture land since the early 1990s (G. Kanapek, area farmer, personal communication). WS personnel have observed and reported feral hog damage in the area since 1995, when they began providing assistance with hog damage management. However, hog activity and

damage varied from year to year based on factors such as weather and area crop production (type and acreage of crops planted). During relatively dry years (i.e. rainfall < 30 in [76.2 cm]), hog activity generally increased in the area because of water availability at the site. During relatively wet years (i.e. rainfall > 30 in), hog activity usually decreased. Also, hog activity and damage was related to the timing of annual rainfall. Annual changes in the types and quantity of crops produced in and around the area influenced the local hog population density. Damage became widespread beginning in the mid-1990s, prior to the beginning of the project.

Since the project began, most of the damage reported in the area has been to corn and grain sorghum crops. However, damage to pasture land and other property has occurred. When WS began feral hog management efforts in 1995, much of the damage occurred in grain fields adjacent to the WMA. Habitat on the WMA provided dense cover and a permanent water supply for hogs in the area. Hogs fed in the grain fields from late afternoon until mid morning and retreated to dense cover during the heat of the day. Hog damage usually began in June and continued until crops were harvested. This damage was sometimes difficult to detect because hogs often traveled to interior areas of grain fields to feed. In some situations, damage was not detected until harvest. Area farmers and WS personnel observed that when hogs began damaging crops at a site, damage generally continued until control measures were initiated.

## HISTORY OF MANAGEMENT EFFORTS

Area landowners and farmers began feral hog management efforts in the early 1990s and initially, producers used multiple catch live traps, shooting, and dogs to manage damage. However, frequently hogs came from the adjoining WMA where access was not allowed. To address the hog damage problem, TPWD allowed limited live trapping by authorized trappers and public hunting with archery equipment on the WMA, but few hogs were removed from the area and feral hog damage continued to be a serious problem. Public hunting for feral hogs continues today on the public lands, but little hunting is done during the summer months when crop damage occurs. Public hunting during summer months on the WMA is restricted to archery equipment and probably has had little impact on feral hog densities and damage in the area. At least one study on public hunting for feral hogs has shown that it had little effect on the feral hog population. Recreational hunting in the Northern Territory of Australia is considered to have no effect on the feral hog population (Choquenot et al. 1996).

Farmers encountered several problems when management was attempted in their grain fields. First, hogs were difficult or impossible to spot in grain fields because of the dense cover the crops provided, which eliminated the option of shooting hogs in the fields. Second, many grain fields were located adjacent to heavy cover associated with bottomlands or overgrown pastures. Third, because it was likely that dogs would pursue hogs into areas where access was not allowed, farmers were reluctant to use this tool.

## DEVELOPMENT OF A MANGEMENT PLAN

In 1995, area farmers and land management officials requested feral hog damage management assistance from the WS Program. In the spring of 1995, WS personnel, resource managers, farmers and local elected officials held a public meeting to discuss developing a feral hog management strategy for the area. The meeting included a discussion of feral hog damage in the project area, management efforts attempted to date, obstacles limiting management efforts, strategies for managing hog damage, and additional efforts and methods which could improve hog damage management operations in the area. Local farmers and landowners held a variety of opinions and concerns about the area's hog problem. Some local residents felt the problem was the result of mismanagement of the WMA, which had created ideal hog habitat. Others in attendance felt the problem could be resolved through an intense lethal management program conducted on the WMA. Other individuals who were interested in hunting feral hogs attended the meeting to offer their assistance with hog control.

The following management methods were discussed: multiple catch live trapping, shooting, hunting with dogs, snaring, exclusion (fencing), habitat management (grazing and mowing to reduce cover), aerial hunting, and monitoring the management area for feral hog activity. Strategies considered included a) a lethal management only approach involving a variety of lethal methods such as trapping and shooting, b) a non-lethal only approach involving exclusion and habitat modification, and c) an IPM approach involving a

combination of all lethal and non-lethal methods considered practical for the area. The IPM approach involving use of a variety of methods was determined to be most appropriate for the area. WS personnel encouraged a cooperative effort for implementing the feral hog management plan. This effort involved integrating local land owners' activities on their properties and allowing WS to conduct feral hog management on the WMA and surrounding private lands.

From its onset, WS employees involved with the program believed that one of the most important aspects would be gaining a thorough knowledge of the area. Employees referred to maps, spoke with landowners and land managers and surveyed the project area to become familiar with the site. WS worked diligently to secure access to and survey as many properties as possible and this effort is continued annually. Familiarity with the area has been an important factor related to quickly resolving feral hog damage problems. As a result of the time spent scouting the management area, personnel are familiar with hog travel routes, preferred escape areas, properties where hogs can be pursued and the best areas in which to conduct management activities. This knowledge has allowed employees to promptly respond to damage situations without the delays associated with surveying and securing access permission.

When the program began, many area farmers were using multiple catch live traps and shooting to remove hogs from the area and they were encouraged to continue these efforts. WS personnel met with area farmers and landowners to demonstrate and recommend the use of neck snares for feral hog capture. Farmers agreed to conduct management efforts when and where they could. Also, the WMA manager agreed to construct a net wire fence around the perimeter of the WMA, mow fence lines, and increase cattle stocking rates to reduce cover for hogs. WMA and US Corps of Engineers managers agreed to allow WS personnel to conduct feral hog management activities on the WMA. Many farmers and landowners in the area also gave WS employees permission to conduct management operations on their properties. The result was the implementation of a cooperative feral hog damage management plan. To date, WS personnel have been granted access to more than 4,800 ac (3,156 ha) of private property surrounding the WMA. An important component to the feral hog management plan was access to areas where hogs could be found or areas where they might retreat during management activities.

## IMPLEMENTING THE MANAGEMENT PLAN

After development of a management plan in 1995, WS personnel began conducting feral hog management activities such as shooting, snaring and hunting with dogs. As the program progressed they identified some measures that improved hog management in and around grain fields. With the exception of a single project in 2005, WS hog management projects were conducted during the grain growing season. Because feral hog damage usually continued at a site until management measures were initiated, a prompt response to each damage situation was of primary concern. Additionally, the strategy within the management plan included methods and efforts to reduce the likelihood feral hogs would disperse from one site to another site. We believe the current program meets these objectives.

Fencing became a key component in the management plan because it permanently impacted feral hog travel to and from thick cover on the WMA. Fencing improved the effectiveness of snaring, restricted feral hog travel and helped field personnel quickly locate hog travel routes. As a result of fencing, hogs traveled through water gaps or locations where the fence had been raised off of the ground. Hogs are capable of breaking through the fence at any location, but rarely do so. Today, approximately 90% of the perimeter of the WMA is fenced with net wire. It has taken several years to reach this point, but fencing has been an important part of the management plan and each extension has improved overall effectiveness. Additionally, area farmers have recently begun to use electric exclusion fences on the perimeters of fields in areas where hogs have been most active. They have reported that the electric fences have further reduced feral hog damage. In some cases, the effectiveness of dogs has been enhanced by fencing because WS employees could more easily determine escape routes and fencing delayed the escape of feral hogs.

Habitat modification has contributed to the overall effectiveness of the management plan because it has eliminated cover for feral hogs in the area. The elimination of cover has improved the results of feral hog shooting and hunting with dogs. Also, it has resulted in less time spent locating feral hogs. When management operations began in 1995, livestock grazing was limited to 983 ac (410 ha) on the WMA. Currently, 5,965 ac (2,413 ha) are grazed by cattle and large areas of dense cover

have been eliminated. As a result of the elimination of cover, it has been much easier to spot hogs when they travel through the area. Establishment of cleared or open areas around the perimeter of grain fields improved shooting operations and allowed for easier inspection of grain fields.

On one occasion, hog damage became so severe in a corn field that the WS employee who worked the site requested that two intersecting swaths be cut across the field to allow for easier access with dogs and to open shooting lanes. The action allowed the WS employee to quickly resolve the problem.

Dogs have also been a key part of the management plan. Often, the quickest way to stop damage in grain fields has been to work the area with dogs. Their greatest benefit has been the secondary harassment of feral hogs and subsequent movement away from fields where damage was occurring. The use of dogs has been limited because some landowners would not allow access to their properties.

Shooting is a versatile tool that has been used under a variety of circumstances. In the area, WS personnel have used firearms to take hogs over baited sites and at wallows or watering sites. Soured grain or other baits have been used to attract hogs to shooting areas. Personnel have also shot hogs as they passed fence crossings. Recently, in an effort to introduce new technology into the management plan, employees have begun using noise suppressed rifles and night vision equipment to take feral hogs. In some cases, personnel have stalked feral hogs at night as they fed or traveled through locations in the area. Prior to the procurement of these technologies, many of the shooting operations were performed during daylight hours when hogs are least active. During dry periods, personnel hunted artificial water sources scattered throughout the area. These sites were prime shooting sites when water was limited. Area farmers and landowners have also shot feral hogs when they had the opportunity. Shooting was often most effective after crop harvest began because hogs traveled over greater distances to locate food, and cover was reduced as harvest progressed.

Snares have also been an important part of the overall management plan. Just as net wire fences make snares more effective, snares make fences a more effective barrier because hogs quickly become reluctant to cross fences when snares are used regularly in an area. WS employees have encountered hogs that avoided snares placed in the area and some hogs have gone to great lengths to avoid areas where snares have been used. Area landowners also use snares to take feral hogs in the area.

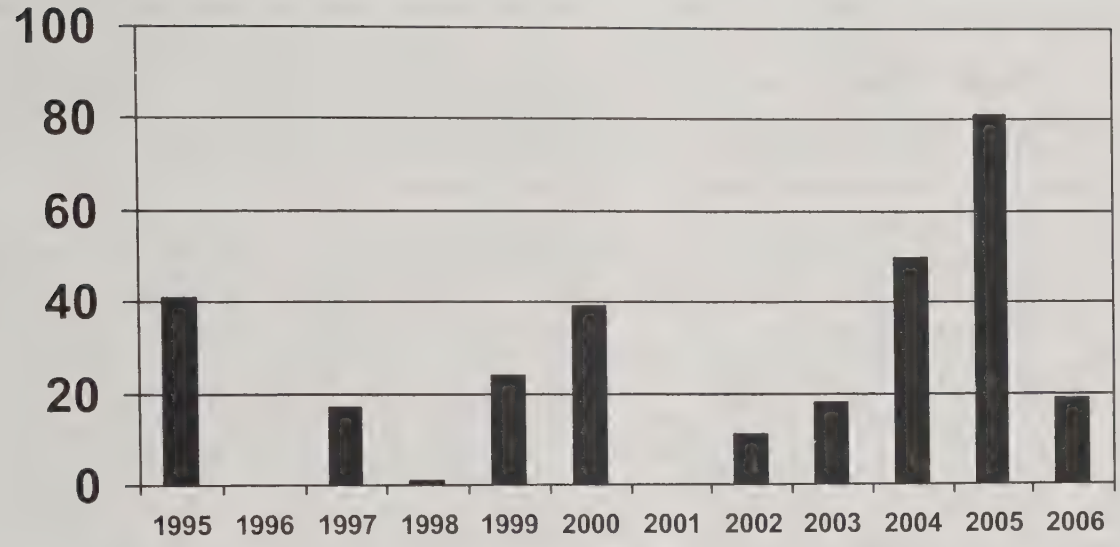
Area farmers and landowners have trapped feral hogs as part of the management plan. Farmers have used multi-catch live traps of various designs to capture hogs in the area. Traps have been least effective during the times when crop damage occurred because of an abundant food supply. WS personnel occasionally used traps in the area, but area farmers and landowners conducted most of the live trapping operations.

All terrain vehicles (ATVs) and off road vehicles have been an important part of feral hog management in the area. These vehicles allowed personnel to quickly access areas that would otherwise have been difficult to access. The vehicles also allowed personnel to ferry equipment and dogs to remote areas. ATVs were beneficial in monitoring the area for hog activity.

In 2005, WS personnel conducted an aerial shooting effort at the project site because farmers and landowners reported that there were a large number of hogs in the area and they were hopeful the operation would reduce damage. WS had never conducted aerial shooting at the site because vegetation was too dense to locate feral hogs during spring and summer months. The aerial shooting operation was conducted in mid-February, prior to the leafing out of vegetation. The operation resulted in the removal of 36 feral hogs from the area and allowed for an aerial survey, which was helpful in learning more about the area.

## RESULTS

Since 1995, WS personnel have removed 301 feral hogs from the area (Figure 1). The number of hogs removed by farmers and local hunters during this period is unknown, but they have occasionally reported some of their hog take and we estimate that they have taken at least as many hogs as WS personnel. In 1996 and 2001, WS did not take feral hogs in the area because farmers did not request assistance following weather related events that affected crop production. In 1998, WS assistance was very limited due to a personnel shortage and only 1 hog was taken. Snares accounted for the largest number (n=105) of hogs taken during the 12-year period. The hog take for the remaining methods during the period was as follows: dogs (n=84), shooting (n=48), aerial shooting (n=36),




| | 1995 | 1996 | 1997 | 1998 | 1999 | 2000 | 2001 | 2002 | 2003 | 2004 | 2005 | 2006 |
|---|---|---|---|---|---|---|---|---|---|---|---|---|
| Snares | 7 | | 4 | 1 | 18 | 15 | | | 13 | 17 | 17 | 13 |
| Cage Traps | | | | | | 2 | | 10 | 2 | | 6 | |
| Shooting | 2 | | | | 3 | 8 | | 1 | | 16 | 18 | |
| Night Vision | 1 | | 1 | | | | | | | 1 | | 5 |
| Dogs | 31 | | 12 | | 3 | 14 | | | 3 | 16 | 4 | 1 |
| Aerial Hunting | | | | | | | | | | | 36 | |

**Figure 1.** Hog take in an area of central Texas by method, 1995-2006.

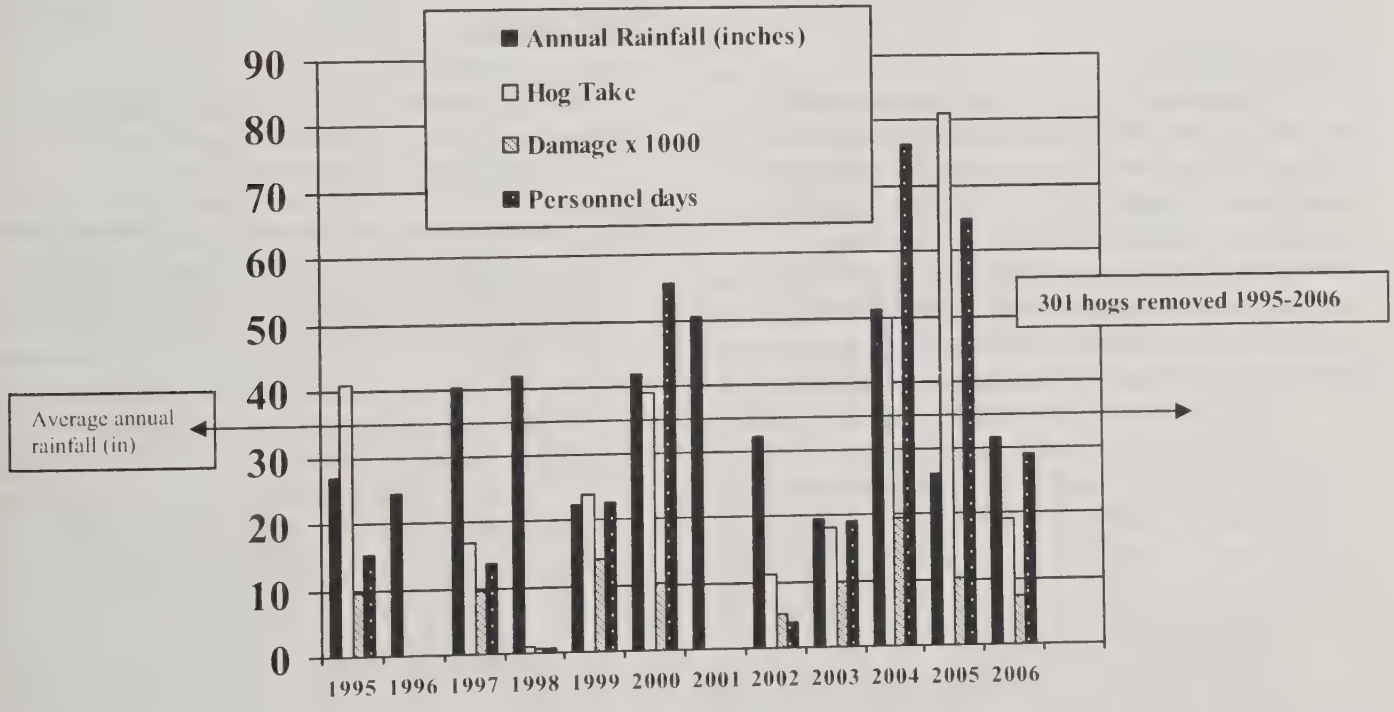



**Figure 2.** Project site annual rainfall, WS feral hog take, crop damage reported by area agriculture producers, and personnel days worked by year for an area of central Texas, 1995-2006.

cage traps (n=20), and shooting with night vision equipment (n=8). The 36 hogs taken during the 2005 aerial shooting operation were taken in 4.4 hrs of flying time.

Feral hog damage reported by area farmers ranged from no damage reported in 1996 and 2001, when WS personnel did not conduct management projects, to $19,500 in 2004. As previously stated, WS personnel usually observed more hog activity in the project area during relatively dry years and less hog activity in relatively wet years. Figure 2 is a comparison of rainfall, farmer reported hog damage, hogs taken, and personnel hours by year. The years 2000 and 2004 received above average rainfall (41.82 in [106.22 cm] and 51.29 in [130.28 cm] respectively) and hog take was high (n=39 and n=50, respectively) compared to most other years. However, these years followed the two driest years during the study period, and in each case, dry conditions did not improve until after crops were planted.

Annual WS assistance (personnel days) with feral hog management at the site ranged from 0 days in 1996 and 2001, when personnel did not conduct operations, to 76.25 days in 2004. Number of WS personnel assigned to annual operations varied from year to year based on personnel availability and feral hog activity at the site.

## CONCLUSION

The objective of the feral hog management plan was to minimize feral hog damage as it occurred in the project areas. WS personnel and local property owners used a variety of methods to manage feral hogs at each project site. At times, lethal hog management operations became less productive because hogs changed their behavior to avoid snares, shooting, dogs and other lethal methods. However, this behavioral change was a benefit to the feral hog management program because it was much easier to move hogs away from damage sites. Feral hog management programs must be evaluated against established objectives (Choquenot et al. 1996). We have found, as have others, that it is preferable for these objectives to be set in terms of damage reduction rather than changes in feral hog numbers because reducing feral hogs may not always lead to acceptable levels of reduction in damage caused by hogs (Choquenot et al 1996). There are situations where significant feral hog population reduction may produce the desired management effect. However, many situations require a different management approach. Given the growing feral hog populations found in many areas, the limited resources available to deal with the problems, and seasonal and annual variations in feral hog densities and activity, the management plan discussed in this paper may be effective in addressing feral hog damage problems in other areas.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

CHOQUENOT, D., J. MCILROY, T. AND KORN. 1996. Managing vertebrate pests: feral pigs. Bureau of Resource Sciences, Australian Government Publishing Service, Canberra, Australia.

LITTAUER, G.A. 1993 Control techniques for feral hogs. Pages 139-147 *in* C. W. Hanselka and J. F. Cadenhead, editors. Feral swine: a compendium for resource managers. Texas Agricultural Extension Service, Kerrville, Texas, USA.

# LANDSCAPE GENETICS OF FERAL SWINE AND IMPLICATIONS FOR MANAGEMENT

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***Abstract:*** The management of the invasive feral pig (*Sus scrofa*) has been the subject of intense study in recent years. Feral pigs are also susceptible to diseases (e.g., brucellosis, pseudorabies) that can be transmitted to livestock, humans, and wildlife. Feral pigs clearly represent a threat to the sustainability of multiple agriculture products. Population reduction (trapping or shooting) is the best current alternative for controlling pig damage. However, reduction is crude and inefficient in terms of manpower and resources because pigs from neighboring areas quickly recolonize managed areas. We used a panel of 9 microsatellite loci to study broad-scale population structure in feral pigs from south Texas and to evaluate recolonization after a local removal. At a broad scale (>200 km), pig populations displayed a moderate degree of genetic structure (Fst = 0.16), suggesting that at broad geographic scales, populations are differentiated enough to be functionally independent. However, genetic similarity was not a simple function of geographic distance, implying that movement and dispersal are not equal among populations. This may be due to the presence of terrain features that promote (e.g., river systems) or inhibit (e.g., urban areas, farmland areas) dispersal. At a local scale, animal samples taken before and after a removal event were genetically different (Fst = 0.08), indicating rapid recolonization occurred into the controlled area. Overall, our results indicate that knowledge of population structure in south Texas could be used to improve pig control efforts, but high rates of movement and dispersal in other areas would likely require control efforts over a very broad region, possibly an entire watershed. Ongoing efforts will attempt to identify fine-scale genetic structure and landscape features that could be used to focus management efforts.

***Key Words:*** damage management, feral pig, genetics, invasive species, population management, *Sus scrofa.*

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Feral pigs (*Sus scrofa*) are considered an exotic invasive in the United States, where conservative estimates indicate an annual loss of $200/pig due to agricultural damage. Feral pigs may have dramatic effects on native ecosystems by excessive rooting and wallowing (Taylor 1993). Feral pigs compete with and predate upon wildlife and livestock species (Synatzske 1979, Taylor 1993, Tolleson et al. 1993, Gipson et al. 1998, Kammermeyer et al. 2003). Finally, feral pigs are also susceptible to diseases that can be transmitted to livestock, humans, and wildlife, including pseudorabies, swine brucellosis, bovine tuberculosis, vesicular stomatitis, and leptospirosis, as well as foreign animal diseases, such as foot and mouth disease, rinderpest, African swine fever, or classical swine fever (Witmer et al. 2003).

Feral pigs clearly represent a threat to the sustainability of multiple agriculture products and natural resources. Population reduction (trapping or shooting) is the best current alternative for controlling pig damage. However, reduction is inefficient in terms of manpower and resources because pigs from neighboring areas quickly re-colonize managed areas. In addition, trapping and shooting methods have a reduced success over time and limited area of population impact (Mapston 2004). Increased knowledge of pig movements and population structuring could dramatically improve the efficiency of management efforts by focusing manpower and resources where they are most effective. For instance, landscape features that influence pig movements could be incorporated into control efforts. Targeting dispersal corridors could prevent re-colonization of managed areas or natural boundaries could be used in a "divide and conquer" strategy. In other cases, the landscape could be altered to prevent immigration.

Furthermore, the scale of management could be adjusted to match the entire population of interest, perhaps by encouraging the formation of management cooperatives in areas where the scale of the population exceeds the average size of landholdings.

Traditional wildlife approaches, including telemetry, tagging, etc., are informative, but costly and limited by sample size and study duration (DeYoung and Honeycutt 2005). Population genetic approaches consider the numerous demographic and stochastic factors affecting population structure and connectivity in a well-developed theoretical framework. However, the application of population genetic theory to wildlife management problems was long limited by a lack of suitable genetic markers and the time and cost-intensive nature of the laboratory methods (DeYoung 2007). A suite of technological advances during the past two decades have resulted in dramatic gains in the number and types of molecular markers, automation of laboratory instrumentation, and increased computer processing speed. Therefore, large-scale genetic studies of wildlife populations are now feasible and offer a powerful new tool for wildlife management (DeYoung and Honeycutt 2005, DeYoung 2007). For instance, the combination of genetic data with geographic information systems (GIS) provides the capability to directly assess the influence of landscape features on population structure and rates of gene flow (Manel et al. 2003).

We are using a panel of microsatellite DNA loci in a landscape-genetic approach to study broad-scale population structure in feral pig populations in south Texas. Our preliminary data indicates that at a broad geographical scale, pig populations display a moderate degree of genetic structure, suggesting that disjunct populations are differentiated enough to be functionally independent. However, genetic and geographic distances among populations do not follow a simple linear relationship. Some geographically proximate populations are genetically dissimilar, while some geographically distant populations are similar. Consequently, movement and dispersal are not equal among populations. Patterns of genetic similarity or dissimilarity among populations are correlated to the presence of terrain features that promote (e.g., river systems) or inhibit (e.g., urban areas, farmland areas) dispersal. Overall, our preliminary results indicate that knowledge of population structure in south Texas could be used to improve pig control efforts. Ongoing efforts will attempt to identify fine-scale genetic structure and landscape features that could be used to focus management efforts.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

DeYoung, R. W. 2007. Genetics and applied management: using genetic methods to solve emerging wildlife management problems. Pages 317-336 *in* T. E. Fulbright and D. G. Hewitt, editors. Frontiers in wildlife science: linking ecological theory and management applications. CRC Press, Boca Raton, Florida, USA.

DeYoung, R. W., and R. L. Honeycutt. 2005. The molecular toolbox: genetic techniques in wildlife ecology and management. Journal of Wildlife Management 69:1362-1384.

Gipson, P. S., B. Hlavachick, and T. Berger. 1998. Range expansion by wild hogs across the central United States. Wildlife Society Bulletin 26:279-286.

Kammermeyer, K., J. Bowers, B. Cooper, D. Forster, K Grahl, T. Holbrook, C. Martin, S. McDonald, N. Nicholson, M. VanBrackle, and G. Waters. 2003. Feral hogs in Georgia: disease, damage and control. Georgia Department of Natural Resources, Wildlife Resources Division, Game Management Section, Social Circle, Georgia, USA.

Manel, S., M. K. Schwartz, G. Luikart, and P. Taberlet. 2003. Landscape genetics: combining landscape ecology and population genetics. Trends in Ecology and Evolution 18:189-197.

Mapston, M. E. 2004. Feral hogs in Texas. Texas Cooperative Extension, Texas A&M University Press, College Station, Texas, USA.

Synatzske, D. R. 1979. Status of the feral hog in Texas. Texas Parks and Wildlife Department, Artesia Wells, Texas, USA.

Taylor, R. B. 1993. History and distribution of feral hogs in Texas. Pages 17-27 *in* C. W. Hanselka and J. F. Cadenhead, editors. Feral swine: a compendium for resource managers: proceedings of a conference. Texas Agricultural Extension Service, College Station, Texas, USA.

TOLLESON, D., D. ROLLINS, W. PINCHAK, M. IVY, AND A. HIERMAN. 1993. Impact of feral hogs on ground-nesting gamebirds. Pages 76-83 *in* C. W. Hanselka and J. F. Cadenhead, editors. Feral swine: a compendium for resource managers: proceedings of a conference. Texas Agricultural Extension Service, College Station, Texas, USA.

WITMER, G. W., R. B. SANDERS, AND A. C. TAFT. 2003. Feral swine-are they a disease threat to livestock in the United States? Proceedings of the Wildlife Damage Management Conference 10:316-325.

# DNA DETECTION OF FOXES TO PREVENT ESTABLISHMENT IN TASMANIA

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***Abstract:*** The red fox (*Vulpes vulpes*) has wreaked havoc on mainland Australia's environment and agricultural production since its introduction in the 1870s. Over the same period, the southern Australian island State of Tasmania has remained virtually fox-free, allowing its unique biodiversity to remain relatively pristine. Recently, an unknown number of foxes were deliberately or accidentally introduced to Tasmania. Some of those animals and possibly their progeny now live in the wild in Tasmania. Finding foxes in a state the size of Tasmania presents special problems for wildlife managers, but is essential to prevent their establishment in this stronghold for Australian marsupials. To assist in finding foxes in Tasmania, we have developed DNA detection approaches specifically for foxes that utilize the ubiquitous mitochondrial DNA found on the surface of mammal scats. Using these approaches, fox DNA has been detected in three different regions in Tasmania and have provided the basis for intense control efforts in those areas. We are now expanding our approach to include other predatory mammals of interest (including both native marsupials and other introduced mammals) and increasing the breadth and scope of our surveys.

***Key Words:*** eradication, DNA-based detection, invasive species, mitochondrial DNA, red fox, scat surveys, Tasmania, *Vulpes vulpes*.

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## INTRODUCTION

Biological invasions have been a major cause of worldwide species endangerment and extinction during the 19th and 20th centuries (Caughley 1994). Island fauna in particular have suffered heavily when exposed to novel competitors or predators (Corke 1987, Burbidge and Manly 2002). Australia, owing to its long isolation is sometimes viewed as an island continent, and has a highly distinctive biota, but it has the worst record for mammalian extinctions globally, with 27 recorded since 1780 (Short and Smith 1994). A major cause of species loss has been the predatory red fox (*Vulpes vulpes*) which was introduced to Australia from Europe in 1855 for hunting, and is now a devastating invasive pest to both wildlife and agriculture (Short and Smith 1994). Foxes have been implicated in the decline of several mainland and island species of Australian mammal, and are listed as a threatening process under the Australian Environment Protection and Biodiversity Conservation Act 1999.

In contrast to most of Australia, the native fauna of Tasmania, a large island state of 68,332 square km, some 200 km to the south of the Australian mainland, is relatively unchanged by European settlement and has seen the known extinction of only one mammal in historical times (the thylacine). Tasmania's isolation has meant that it has been spared predation by the fox and, as a consequence, now represents a living repository of pre-European Australian marsupial communities that is unparalleled elsewhere.

In September 2001, a fox was shot in northern Tasmania and there were reports of a second shot in the midlands, near Conara. In 2003, the fresh remains of a fox were found near Burnie on Tasmania's northern coast and two more carcasses (one at Lillico Beach near Devonport in December 2005 and a second at Cleveland/Conara in the midlands in August 2006) have been found (Figure 1). These foxes are believed to have been part of, or the progeny of, a deliberate introduction of an unknown number of foxes to the island around 1999/2000 (Saunders et al. 2006). The risk that this introduction poses to Tasmania's wildlife (Saunders et al 2006), and its tourism and pastoral industries led to the establishment of a taskforce by the Tasmanian and Australian federal governments to define and manage the threat.

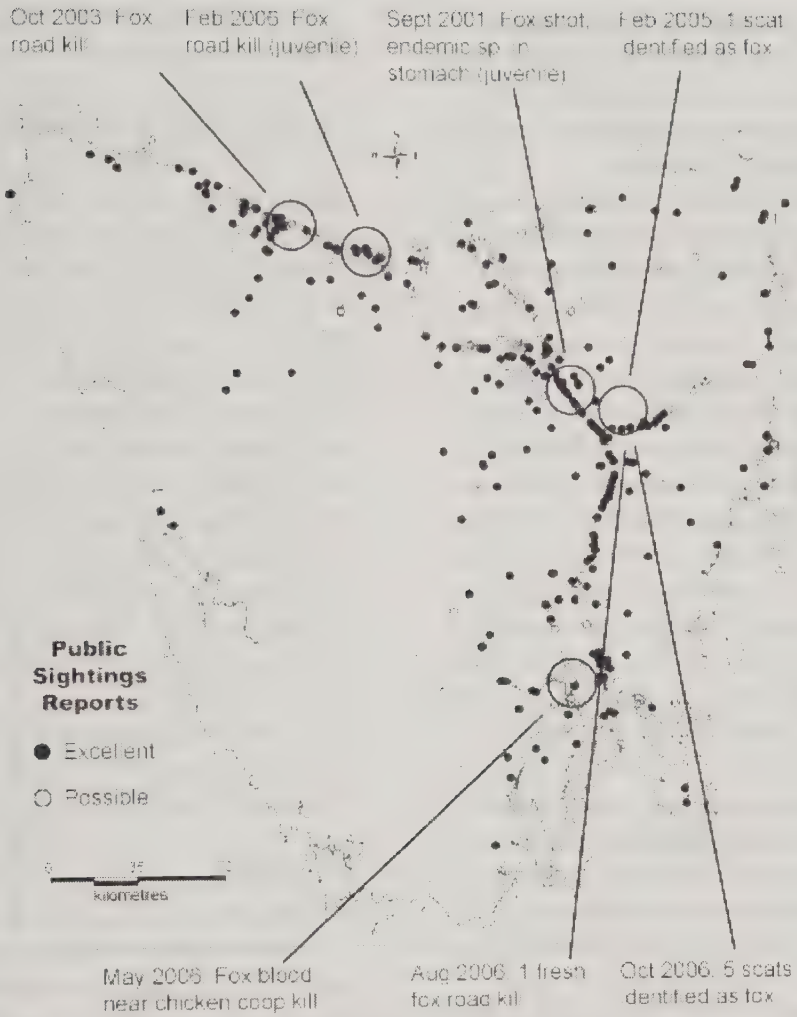



**Figure 1.** Fox sightings reports in Tasmania since 2001 and the sites of scat identified as positive by DNA testing. Fox sightings are focussed around centres of human habitation and along major highways which is likely to present a bias in sampling rather than a true indication of fox activity. Proposed sampling for scats in the future will be directed at a systematic collection across the state.

## TACTICAL MONITORING APPROACHES

Over 1,500 fox sightings have been reported by the public since 2002 (Figure 1), suggesting that the fox may already be widespread on the island. However, most reported sightings have been difficult to verify, and many are believed to be erroneous. This lack of unequivocal evidence has led to public scepticism about the threat posed by foxes and consequent political pressure to reduce funding on attempts to monitor and control the invasion.

The control program implemented by the Tasmanian Fox Eradication Program has been directed towards tactical baiting with the poison 1080 (sodium monofluoroacetate) which can be toxic to native fauna. In order to minimise non-target deaths (particularly given the rich fauna of medium sized marsupial carnivores), baits must be set by hand, which is expensive and time consuming. Currently, baiting is centred on regions where reported sightings of foxes have been most concentrated or where hard evidence (e.g., road-kill carcasses) has been found. Accurate detection and monitoring of the fox is critical to the efficacy of such control operations. However, given the likely high incidence of error, the dispersed nature of the sightings and the fact that they are concentrated around regions of high human activity such as major highways, this targeting can be at best approximate and its effectiveness difficult to determine.

The biased and uncertain nature of public sightings requires that other means of monitoring this elusive predator be developed. One approach that does not rely on ad hoc, fleeting and usually

nocturnal sightings, is the collection of predator scats (faeces). Canid scats can persist in the field for weeks or even months (Kohn et al. 1999), and several species, including foxes, tend to leave them in prominent places such as along trails (MacDonald 1980). To bolster evidence from reported sightings, earlier phases of this program employed analysis of faecal (scat) morphology and hair to define fox presence. The main advantages of such surveys is that they can be done by relatively untrained personnel and provide exact location data for targeted control. The disadvantage is that definitive morphological diagnosis can be difficult in a place like Tasmania where there are at least 6 major mammalian predators that can produce scats of similar size and shape. Diagnostic hairs associated with scats, can provide additional information about the species of origin, but occur in only about 10% of scats.

## DEVELOPMENT OF A DNA TEST FOR FOXES

DNA analysis of scats provides a solution to this problem, by combining the benefits of sampling offered by scats with the robust detection provided by approaches based on the polymerase-chain-reaction (PCR, Kohn and Wayne 1997). These approaches have been primarily used to study the ecology of native wildlife. However, the same principles can be applied to the detection and mapping of elusive invasive species, such as the introduced red fox. With appropriate verification of the methodology ((Taberlet and Luikart 1999), non-invasive DNA-based methods could provide the high-quality distribution data that is required for effective control.

To this end, we have developed a PCR-based test specific to foxes that excludes the amplification of other carnivore DNA and provides a rapid initial screen of all scats collected. Further verification is required following the initial identification of a scat positive for fox DNA. The full details of this test and its development can be found in Berry et al. (2007). The key points of the test and its development are as follows.

1. The test involves a single multiplex PCR that amplifies fragments from two genes of the mitochondrial DNA genome. The first fragment targeted is a portion of the 12 rRNA gene, and is designed to amplify in the presence of DNA from any carnivore present in Tasmania. The amplification of this fragment acts as a control for the successful extraction of DNA from the scat. The second fragment amplified is from the cytochrome b gene and targets fox DNA specifically.
2. Field testing demonstrated that our approach to detecting fox DNA in scats was 100% successful in scats up to 12 weeks old despite significant loss of template DNA through degradation. It is quite likely that a high rate of detection is possible for scats that are considerably older than 12 months.
3. The test was implemented by the Department of Primary Industry and Water in 2004 for scat identification activities on contract to the Wildlife Genetics Laboratory at the University of Canberra. Following the initial implementation, additional analytical procedures were added to the test to minimise the risk of false positive identifications of fox traces (see below).
4. The positive identification of scat or other trace material now acts as a trigger for the focus of fox eradication actions.

## IMPLEMENTATION OF THE DNA TEST FOR TASMANIAN FOX ERADICATION

The implementation of the test for samples derived from Tasmania necessitated the development of a series of protocols to maximise the value of each sample collected and to minimise the risk of Type 1 errors (false identification of a sample as fox when it is not) or Type 2 errors (failure to detect a fox scat). The minimisation of these risks falls into three general categories: (1) sample handling methods that minimise the loss of DNA from a sample once it had been collected and, therefore, minimise the risk of not identifying a scat as from fox when it is one (Type 2 error), (2) strategies that minimise the risk of sample contamination from collection to analysis (Types 1 and 2 errors), and (3) strategies that minimise the risk of false positives for foxes (Type 1 errors). Below, we record the approaches taken to minimise these risks. A fourth category of risk minimisation relates to the probability of a fox scat not being found should it actually be in an area surveyed and the probability that an area containing fox scats is not searched at all. Given the immense area of suitable habitat for foxes in Tasmania (>30,000 $km^2$) the risk of not searching an area containing foxes is substantial and its minimisation requires a strategic and systematic approach to survey. That

approach is currently under development and will not be detailed here.

**Sample Handling to Minimise the Loss of DNA from a Sample**

DNA in the epithelial and other cells attached to scats will start to deteriorate immediately as cellular enzymes (nucleases) start to degrade the DNA. Large DNA fragments will be cut into smaller and smaller pieces by this process. Two key characteristics of the scat will influence the rate at which DNA degradation occurs are the amount of moisture retained in the cells and the temperature to which the scat is exposed (Murphy et al. 2002). Low moisture content will retard enzyme activity and reduce DNA degradation that may occur through hydrolytic action and cool temperatures will slow the rate of degradation by reducing biological activity. The rate of degradation of the DNA on and within scats can therefore be minimized by removing moisture from the scat and keeping them cool. Therefore, our approach has been to dry the scats as quickly as possible following collection. An alternative approach would be to freeze the scats or preserve them in ethanol, but this makes air transport difficult and expensive (airline companies often do not like transporting dry ice or flammable material) and creates storage problems if the scats are to be stored for a long time. As a consequence we have adopted the following approach:

1. All scats are collected in non-greased paper bags and sealed at the point of collection. This allows air to circulate through and around the scat and begin the drying process.
2. All scats are then dried in a drying room in Launceston, Tasmania, where temperatures are maintained at around 25°C (±5°C).
3. An electric fan and dehumidifier are used to maintain airflow and decrease air moisture in the drying room. Drying takes 2-3 days.
4. Once dry, scats are packed in cardboard boxes and mailed for DNA analysis.
5. Following analysis, scat samples are archived at the CSIRO Australian National Wildlife Collection for future work which may include further species identification or DNA-based dietary analyses.

**Minimising the Risk of Sample Contamination at Collection**

The introduction of foreign DNA (including human) may compromise the ability for the DNA of the target species to be amplified in the laboratory and, hence, reduce the probability that fox DNA is detected in a scat should it be present. As a consequence, it is important that scats are not handled directly by the collectors. There is also a significant risk of disease transfer to scat collectors so occupational health and safety concerns dictate a no handling policy as well. Our approach is that scats must be picked up using either disposable latex gloves or 'chopsticks' improvised from vegetation. New sticks or gloves are used for each scat to minimise the risk of cross-contamination. Scats and animal parts are not allowed to touch personnel, clothing or any equipment other than the sticks or disposable gloves and the interior of the paper bags. Used latex gloves are collected in a bag and disposed of appropriately upon return. Calico bags are used for carrying collected scats and no other material. Other signs (such as fur or blood) collected as part of the activities are stored separately to the scats. Once the scats are bagged, they are never removed from the bag and go directly to the drying room.

**Minimising the Risk of False Positives for Foxes**

The analysis of trace samples of DNA such as those found on animal scats includes a risk of false positives through artefacts of the PCR process. Such artefacts will generally arise from the amplification of closely related DNA sequence from DNA contaminates in the laboratory itself or from prey DNA present in the scat. The potential for laboratory-based contamination is managed rigorously through a series of measures that include the spatial separation of DNA extraction, PCR set-up, post-PCR analysis, the one-way movement of samples and laboratory technicians from DNA extraction stations to the analytical laboratory, operation in DNA-free biohazard safety cabinets, and the inclusion of negative controls at all stages of preparation to test for possible contamination. These approaches follow closely the principles that are used for DNA analysis in human forensic case work. To minimise the risk of contamination by prey items, the primers used to amplify the fox

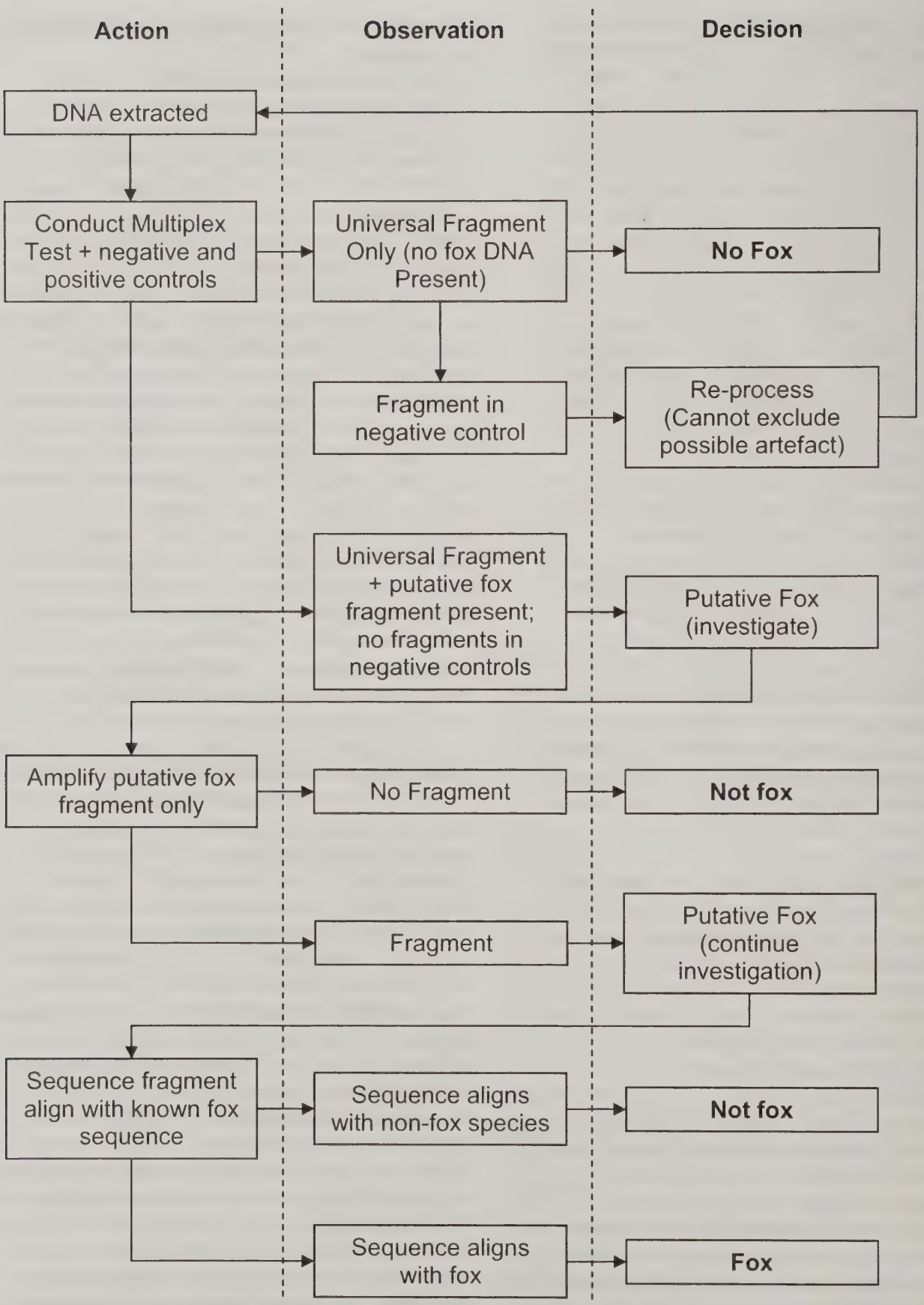



**Figure 2.** Flow diagram showing the sequence of tests applied before a sample can be considered to contain fox DNA. Only a sequence matching those of fox in the absence of fragments in negative controls act as a trigger for control action.

specific cytochrome b fragment in our study were designed to specifically exclude the amplification of homologous sequence from other large carnivores in Tasmania (Berry et al. 2007). However, occasionally, non-target fragments of a size similar to the target fox band are amplified and require investigation. Such fragments are sequenced and compared to sequence contained in the world-wide database GenBank (www.ncbi.nlm.nih.gov/blast/Blast.cgi) and from our own sequence records to determine the species of most likely origin. Only when a sequence from a scat matches specifically to fox do we consider a scat to be positive for fox (Figure 2).

## APPLICATION OF THE FOX TEST IN TASMANIA

We have applied the test to over 2,000 scats and other trace samples collected since 2004 as part of the ongoing program to eradicate invasive foxes from Tasmania. Most scats have been collected as part of a tactical program targeting areas of fox activity as identified through reported sightings from the public. In all, we have identified six scats (from one general region, Figure 1) as containing fox DNA and have also identified a blood sample left at a chicken coop as being from fox. These findings have assisted Tasmanian authorities to target their baiting campaigns by providing solid evidence of fox presence.

A clear limitation of this 'tactical' approach used to date is that it targets areas in a biased fashion, being skewed towards areas in which most people live (and hence are likely to see foxes). In the future, the Tasmanian Department of Primary Industry and Water, the University of Canberra, and the Invasive Animals Cooperative Research Centre will conduct a systematic survey of the state for scats that will maximise the opportunity for finding fox traces and, hence, define the extent of the fox problem in Tasmania. Of central interest is how widespread foxes are. This will be used to guide the planning of strategic eradication activities. If they are widespread, then an assessment of eradication feasibility will be required.

## ACKNOWLEDGMENTS

We thank Craig Bester for producing the map of sightings reports and the Invasive Animals Cooperative Research Centre, the Tasmanian Government, and the Commonwealth Department of Heritage, Environment and the Arts for funding support. The original development of the fox test was supported by the Hermon Slade Foundation and the University of Canberra.

## LITERATURE CITED

BERRY, O., S. D. SARRE, L. FARRINGTON, AND N. AITKEN. 2007. Faecal DNA detection of invasive species: the case of feral foxes in Tasmania. Wildlife Research 34:1-7.

BURBIDGE, A. A., AND B. F. J. MANLY. 2002. Mammal extinctions on Australian islands: causes and conservation implications. Journal of Biogeography 29:465-473.

CAUGHLEY, G. 1994. Directions in conservation biology. Journal Of Animal Ecology 63:215-244.

CORKE, D. 1987. Reptile conservation on the Maria Islands (St. Lucia, West Indies). Biological Conservation 40:263-279.

KOHN, M. H., AND R. K. WAYNE. 1997. Facts from faeces revisited. Trends in Ecology & Evolution 12:223-227.

KOHN, M. H., E. C. YORK, D. A. KAMRADT, G. HAUGHT, R. M. SAUVAJOT, AND R. K. WAYNE. 1999. Estimating population size by genotyping faeces. Proceedings of the Royal Society of London, Series B 266:657-663.

MACDONALD, D. W. 1980. Patterns of scent marking with urine and feces amongst carnivore communities. Symposium of the Zoological Society of London 45:107-139.

MURPHY, M. A., L. P. WAITS, K. C. KENDALL, S. K. WASER, J. A. HIGBEE, AND R. BOGDEN. 2002. An evaluation of long-term preservation methods for brown bear (*Ursus arctos*) faecal DNA samples. Conservation Genetics 3:435-440.

SAUNDERS, G., C. LANE, S. HARRIS, AND, AND C. DICKMAN. 2006. Foxes in Tasmania: a report on the incursion of an invasive species. Invasive Animals Cooperative Research Centre.

SHORT, J., AND A. SMITH. 1994. Mammal decline and recovery in Australia. Journal of Mammalogy 75:288-297.

TABERLET, P., AND G. LUIKART. 1999. Non-invasive genetic sampling and individual identification. Biological Journal of the Linnean Society 68:41-55.

# ECOLOGY OF AN INVASIVE PREDATOR IN HAWAI`I

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***Abstract:*** Cats (*Felis catus*) brought to Hawai`i in the 1700s now occupy most habitats throughout the islands, including montane and sublapine zones. We studied home range, population genetics, diseases, and diet of feral cats on Hawai`i Island. Feral cats on Mauna Kea live in low densities and exhibit some of the largest reported home ranges. While 95% kernel home range estimates for 4 males ( $\bar{x}$ = 1418 ha) were nearly twice as large as 3 female home ranges ( $\bar{x}$ = 772 ha), one male maintained a home range of 2050 ha. Population genetics revealed that Mauna Kea may be a source population for feral cats on Mauna Loa and Hawai`i Volcanoes National Park (HAVO). Assignment tests provided strong evidence for male-biased dispersal from Mauna Kea to Mauna Loa. Mauna Kea cats exhibited high seroprevalence for toxoplasmosis (37.3%) and feline leukemia virus (FeLV; 16.2%) distributed among all age and sex classes. Feline immunodeficiency virus (FIV) occurred only in adult males comprising 8.8% of the overall population. We found that cats on Mauna Kea primarily consumed birds, present in 69% of digestive tracts, whereas birds were in only 28% of samples from HAVO. Within HAVO, prey use differed between Kīlauea and Mauna Loa. On Mauna Loa, more feral cats consumed small mammals (89%), primarily rodents, than on Kīlauea Volcano (50%). Mice (*Mus musculus*) were the major component of the feral cat diet on Mauna Loa, whereas Orthoptera were the major component of the diet on Kīlauea. A digestive tract from Mauna Loa contained a mandible set, feathers, and bones of an endangered Hawaiian Petrel (*Pterodroma sandwichensis*). Abundant birds on Mauna Kea may maintain large numbers of feral cats that disperse long distances. Abundant rodents found in Mauna Loa digestive tracts may also support cats that then take advantage of breeding petrels.

***Key Words:*** diet, endangered species, *Felis catus*, feral cat, genetics, Hawai`i, home range, invasive species, predation.

Managing Vertebrate Invasive Species: Proceedings of an International Symposium (G. W. Witmer, W. C. Pitt, K. A. Fagerstone, Eds). USDA/APHIS/WS, National Wildlife Research Center, Fort Collins, CO. 2007.

## INTRODUCTION

Ecological studies of invasive predators can be important for at least two major purposes: (1) to determine the extent of predation on native and endangered wildlife, and (2) for developing effective control methods (Veitch 1985, Edwards et al. 2000). For example, disease data may help to understand the risk of transmission to other species, diet studies may yield information about formulating attractive baits, home range size and daily movements may inform trap placement and spacing, and population genetics can provide information about the demography of invasive species useful for developing effective control strategies. Despite the long history of feral cats in Hawai`i, there has been little research on their basic ecology or effects on native wildlife. There are also few published accounts of predation by feral cats on endangered species.

Domestic cats were brought to Hawaii on European ships in the late 1700s (King 1984) and apparently established feral populations by 1840 in remote montane areas of Hawai`i Island (Brackenridge 1841). Feral cats were abundant in forests of Lāna`i and O`ahu by 1892 (Rothschild 1893), and predation of forest birds in the late 1800s was reported by Perkins (1903). Where cats have been abandoned in coastal and lowland areas, high density colonies are maintained by feeding (Winter 2003), which causes mortality to nesting colonial seabirds (Smith et al. 2002). However, in remote montane forests and subalpine areas of Maui (Simons 1983) and Hawai`i Island (Hu et al. 2001) feral cats have lived in comparably solitary,

low density situations for more than a century without human subsidies (Tomich 1986). At higher elevations, feral cats are the most important predators of endangered Hawaiian birds including burrow-nesting seabirds (Natividad-Hodges and Nagata 2001), ground-nesting waterfowl (Banko 1992), and tree-nesting passerines (Hess et al. 2004). Cats are also the definitive host of toxoplasmosis, a disease which has killed both endangered birds and marine mammals in Hawai`i (Work et al. 2000, Work et al. 2002, Honnold et al. 2005).

## OBJECTIVES

Our objectives were to determine feline disease prevalence, diet, daily movement rates, home range, and population genetics of feral cats, primarily on Hawai`i Island. We intended to use this information to determine the impacts of feral cats in remote natural areas inhabited by endangered species and to develop a large-scale control strategy. Data presented here came from feral cats that were live-trapped between 1998-2005 on Mauna Kea, and 2000-2005 in Hawaii Volcanoes National Park (HAVO) to reduce predation on endangered Hawaiian birds such as Palila (*Loxioides bailleui*), Nēnē (Hawaiian goose; *Branta sandvicensis*), and `Ua`u (Hawaiian Petrel; *Pterodroma sandwichensis*).

## METHODS AND RESULTS

### Feline Diseases

We determined prevalence to feline immunodeficiency virus (FIV) antibodies, feline leukemia virus (FeLV) antigen, and *Toxoplasma gondii* antibodies in feral cats captured on Mauna Kea Hawaii from April 2002 to May 2004 (Danner et al. 2007). SnapTM Combo FeLV Antigen/FIV antibody enzyme-linked immunosorbent assays (ELISAs) (IDEXX Laboratories, Inc., Portland, Maine, USA), were used in the field with fresh whole blood according to the manufacturer's instructions. Remaining blood samples were sent to Colorado Veterinary Diagnostic Laboratories (Colorado State University, Fort Collins, Colorado, USA) for analysis of antibodies to *T. gondii*. Six of 68 (8.8%) and 11 of 68 (16.2%) cats were antibody positive to FIV and antigen positive for FeLV, respectively; 25 of 67 (37.3%) cats were seropositive to *T. gondii* (Table 1). About 7.5% of cats had recent or active *T. gondii* infections and may have been capable of transmitting toxoplasmosis to other wildlife through oocysts shed in feces. Antibodies to FeLV and *T. gondii* occurred in all age and sex classes, but FIV occurred only in adult males. The cat population on Mauna Kea may be depressed by feline diseases, but the presence of these diseases suggests they would not make effective biological control agents.

**Table 1.** Prevalence (percentage) of feline immunodeficiency (FIV) antibodies, feline leukemia (FeLV) antigen, and *Toxoplasma gondii* antibodies indicating exposure or recent infection in feral cats on Mauna Kea, Hawaii, 2002-04.

| | | | *T. gondii* | |
|---|---|---|---|---|
| | FIV (n = 68) | FeLV (n = 68) | Exposure (n = 67) | Infection (n = 67) |
| Adult | | | | |
| Male | 17 | 17 | 31 | 3 |
| Female | 0 | 13 | 45 | 18 |
| Juvenile | | | | |
| Male | 0 | 33 | 67 | 0 |
| Female | 0 | 17 | 29 | 0 |
| Overall | 9 | 16 | 37 | 7 |

**Diet**

We analyzed digestive tract contents of 143 feral cats from Mauna Kea, 42 from Hawai`i Volcanoes National Park, and 15 from the island of Kaho`olawe to determine diet. We used consistent methods between each of these locations (Hess et al. 2004, In Press). Entire carcasses were frozen, and then stomachs and intestines were removed and preserved in 70% ethanol. Stomachs were cut open along the concave side and intestines were opened along their entire length. A continuous, gentle stream of water was used in combination with gentle pressure from fingers to rinse and separate food items through soil sieves (US Bureau of Standards size 5 and 10 mesh). Prey items were sorted into respective prey types (e.g., small mammals, invertebrates, birds) and identified to lowest taxonomic level with the help of reference collections. The prey type with the highest frequency of occurrence from Mauna Kea was birds, present in 69% of digestive tracts, whereas birds were in only 28% of samples from HAVO (Figure 1). Approximately 27% of cats on Kaho`olawe contained birds, and < 14% contained small mammals, but > 86% contained invertebrates, including some intertidal marine species.

On Kīlauea and Mauna Loa in Hawai`i Volcanoes National Park, small mammals, invertebrates, and birds were the most common prey types consumed by feral cats (Figure 2). Birds occurred in 27.8–29.2% of digestive tracts. We found a significant difference between Kīlauea and Mauna Loa in the total number of bird, small mammal, and invertebrate prey in the digestive tracts of feral cats ($\chi^2 = 8.39$, DF = 2, $P = 0.015$; Hess et al. 2007). On Mauna Loa, significantly more (89%) feral cats consumed small mammals, primarily rodents, than on Kīlauea Volcano (50%) (2-sample t-test; DF = 29, $t = -3.65$, $P < 0.001$). Mice (*Mus musculus*) were the major component of the feral cat diet on Mauna Loa, whereas Orthoptera were the major component of the diet on Kīlauea. We recovered a mandible set, feathers, and bones of an endangered Hawaiian petrel (*Pterodroma sandwichensis*) from a digestive tract from Mauna Loa. This specimen represents the first well-documented endangered bird to be recovered from the digestive tract of a feral cat in Hawai`i and suggests that feral cats prey on this species.

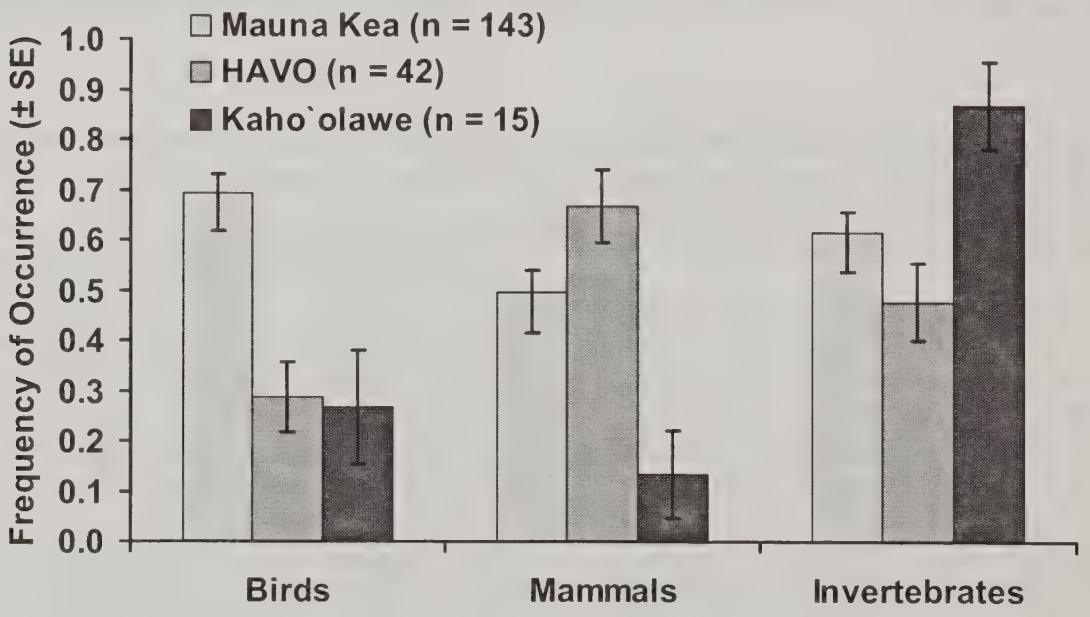



**Figure 1.** Frequency of occurrence (± SE) of three prey types from digestive tracts of feral cats from Mauna Kea, Hawai`i Volcanoes National Park (HAVO), and the Island of Kaho`olawe, Hawai`i 1998–2005.

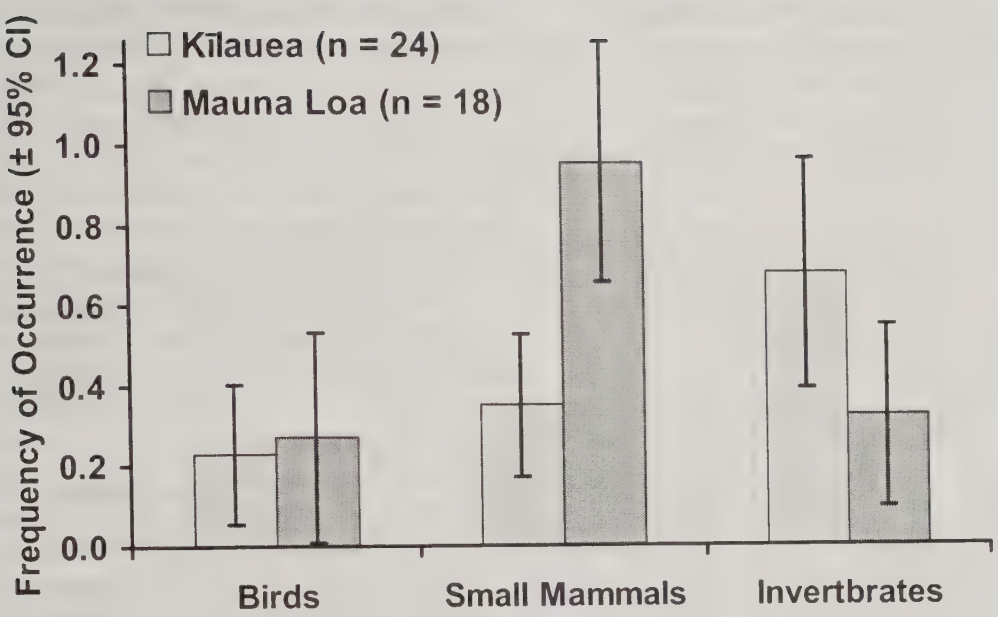



**Figure 2.** Frequency of occurrence of three prey types (±95% CI) from digestive tracts of feral cats from Kīlauea and Mauna Loa in Hawai'i Volcanoes National Park, 2000–2005.

**Daily Movements and Home Range**

We studied the daily movement rates and homes ranges of feral cats on Mauna Kea by capturing and fitting 7 males and 3 females with 35-37g Holohil Systems Ltd. model MI-2 radio transmitters, and tracking these animals between 1999 and 2001. To reduce temporal autocorrelation of observations, we calculated t2/r2 statistics for a range of minimum times between observations and determined a minimum separation of 2.85 days was needed to achieve quasi-independent observations (Swihart and Slade 1985, Swihart and Slade 1986). After excluding data points that were taken less than 2.85 days after the previous observation, we calculated 95%, 50%, and 25% fixed kernel home ranges using the animal movement extension (Hooge and Eichenlaub 1997) for Arcview GIS (ESRI 1999). We used least squares cross-validation to estimate a kernel smoothing parameter for each cat, and used the median value (382) for all cats as the final home range estimate as recommended by Seaman and Powell (1996). We then examined the effect of reduced sample size on kernel home range estimates with 1,000 bootstrap minimum convex polygons (MCP) from the remaining locations using the animal movement extension (Hooge and Eichenlaub 1997). We plotted MCP area against sample size to determine if sufficient observations existed to stabilize MCP area; there were sufficient observations to determine reliable home ranges for 4 of the 7 males and all 3 females. Mean daily movement rates between sexes overlapped widely and did not differ (Unequal variance 2-sample t-test; DF = 6, $t$ = -2.08, $P > 0.08$; Table 2). However, we found that male feral cats on Mauna Kea exhibit some of the largest reported home ranges (Figure 3). While 95% kernel home range estimates for 4 males ( $\bar{x}$ = 1418 ha) were nearly twice as large as 3 female home ranges ( $\bar{x}$ = 772 ha), one male maintained a home range of 2050 ha (Table 3). Log-transformed 95% (Equal variance 2-sample t-test; DF = 5, $t$ = -3.20, $P < 0.03$) kernel home ranges for males were significantly larger than those of females, but 25% kernel home ranges for females were larger than those of males (Equal variance 2-sample t-test; DF = 5, $t$ = 3.53, P > 0.02), indicating that males use extensive areas while females make more intensive use of smaller areas. One male's home range was on the West Slope for five months, then shifted 15 km to the South Slope, and never returned.

**Population Genetics**

We studied seven highly polymorphic microsatellite markers to estimate the genetic structure of three feral cat populations on Hawaii Island. Specific objectives of our genetics research were to (1) evaluate genetic diversity and

**Table 2.** Mean daily movements of feral cats on Mauna Kea, Hawaii 1998-2001.

| Sex | Observations (*n*) | Mean (m/day) | Range (m/day) |
|---|---|---|---|
| Male ($n = 7$) | 572 | 2274 | 91–6014 |
| Female ($n = 3$) | 448 | 135 | 109–184 |
| Overall ($n = 10$) | 1020 | 241 | 91–6014 |

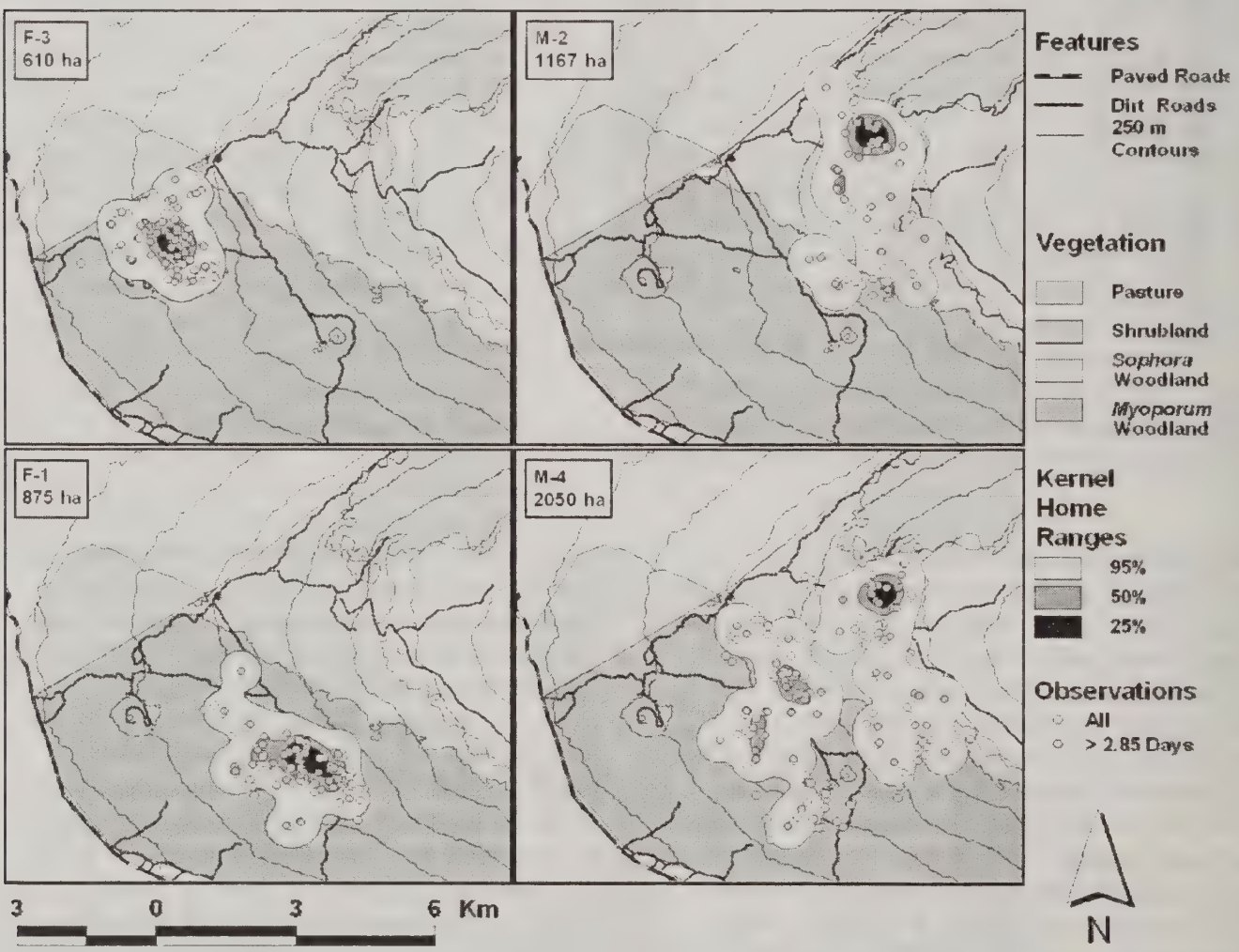



**Figure 3.** Kernel home range estimates (ha) of feral cats on Mauna Kea, Hawai`i 1998-2001. Non-independent observations were eliminated based on estimated average of 2.85 days to quasi-independence.

**Table 3.** Kernel home range estimates (ha) of feral cats on Mauna Kea, Hawai`i 1998-2001. Non-independent observations were eliminated based on estimated average of 2.85 days to quasi-independence. Home range was calculated by the median least square cross validation (LSCV) smoothing parameter (H) value of 378. Asterisk indicates significant differences between sexes at $P < 0.03$ based on 2-sample t-test with equal variances.

| Sex | 25% Kernel* Mean | 95% Kernel* Mean | 95% Kernel Range |
|---|---|---|---|
| Male ($n = 4$) | 32.5 ha | 1418.0 ha | 1167-2050 ha |
| Female ($n = 3$) | 52.7 ha | 772.0 ha | 610-875 ha |
| Overall ($n = 7$) | 41.1 ha | 1141.1 ha | 610-2050 ha |

population structure; (2) assess levels of gene flow and connectivity between populations; (3) identify potential source populations; (4) characterize population dynamics; and (5) evaluate evidence for sex-biased dispersal to formulate an island-wide control strategy (Hansen et al. in press). We collected muscle tissue samples from 85 feral cats (49 males, 36 females) and stored samples in lysis buffer at -20°C until extraction. We extracted genomic DNA from the tissue using the QIAGEN® DNeasy™ Tissue Kit (Qiagen, Inc., Valencia, California, US). Technical detail of extraction, PCR amplification, and analysis can be found in Hansen et al. (In Press).

*Genetic diversity.* High genetic diversity, low structure, and high number of migrants per generation supported high gene flow that was not limited spatially. The mean number of alleles ($A$) ranged from 7.57 ± 2.99 (mean ± SD) to 9.00 ± 3.83 according to population. After Bonferonni corrections, $f$ values showed a significant departure from Hardy-Weinberg equilibrium in NMK and WMK, suggesting that these populations were inbred (Aspi et al. 2006).

*Population structure.* Pairwise estimates of population structure ($F_{st}$), a measure of the genetic variability among populations, were all significant ($P < 0.05$) and calculated (with 95% CI) as 0.038 (HAVO-NMK; 0.008 – 0.071), 0.028 (HAVO-WMK; 0.013 – 0.041) and 0.023 (NMK-WMK; 0.006 – 0.043), indicating gene flow among all populations.

*Gene flow and migration rate.* Migration rates revealed that most migration occurred out of West Mauna Kea. The estimated number of migrants per generation ($Nm$) was 6.3 between NMK and HAVO, 8.8 between WMK and HAVO, and 10.6 between NMK and WMK. The mean posterior probabilities of migration rates showed that the majority of individuals were native to their capture locations in all populations with the most originating in WMK (0.708 – 0.927) (Table 4). There was a relatively high degree of migration between populations from WMK to NMK ($m$ = 0.248; 95% CI = 0.032 – 0.325) and HAVO ($m$ = 0.176; 95% CI = 0.034 – 0.312). In contrast, migration rates from both HAVO and NMK into the other populations were very low with the smallest migration rate from HAVO to WMK ($m$ = 0.015; 95% CI = 0.0004 – 0.048).

*Sex-biased dispersal.* Three measures supported male-biased dispersal and female philopatry (Figure 4). The mean assignment index ($mAI_c$) is a measurement of the likelihood of an individual's genotype to occur in its sampled location. Negative $AI_c$ values indicate dispersal and positive $AI_c$ values indicate philopatry. Relatedness of males and females differed significantly ($P < 0.05$) and the $mAI_c$ of males among populations was significantly lower than that of females ($mAI_c$ = -0.900, 1.23, $P = 0.007$, respectively). The $F_{st}$ value, a measure of genetic variability between males and females, was significantly lower among dispersing males. The higher inbreeding coefficient $F_{is}$ (less inbred) also supported male biased dispersal.

**Table 4.** Migration rates of feral cats between north Maun Kea (NMK), west Mauna Kea (WMK), and Hawai`i Volcanoes National Park (HAVO) on Hawaii Island. Migration rates (with 95% confidence intervals) were estimated as the proportion of individuals in column populations that originated from populations in rows. Values along the diagonal are the proportions of individuals within a population derived from that population.

| | HAVO | NMK | WMK |
|---|---|---|---|
| HAVO | 0.708 (0.667–0.835) | 0.116 (0.004–0.277) | 0.176 (0.034–0.312) |
| NMK | 0.015 (0.0004–0.048) | 0.737 (0.668–0.953) | 0.248 (0.032–0.325) |
| WMK | 0.038 (0.001–0.104) | 0.035 (0.002–0.127) | 0.927 (0.843–0.986) |

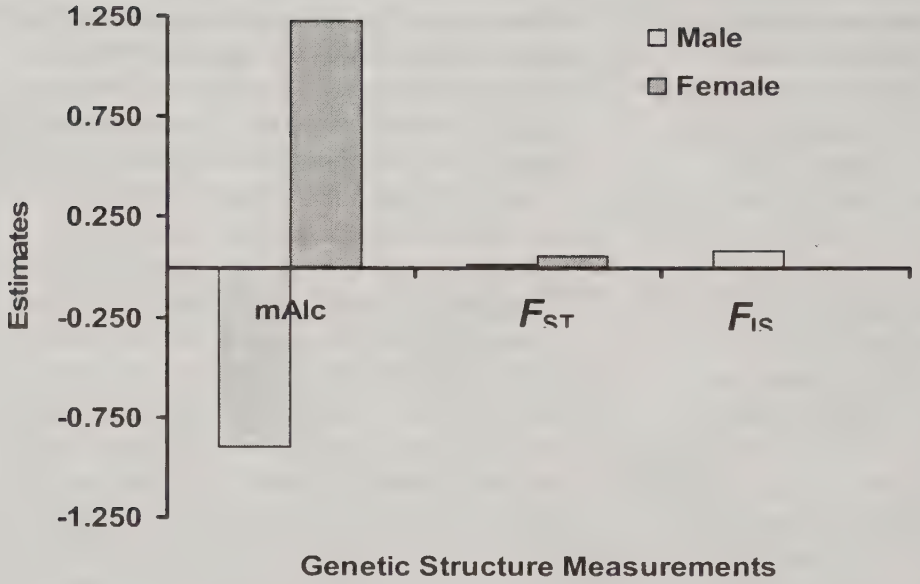



**Figure 4.** Differences in genetic structure measurements between sexes in feral cats on Hawai`i Island. A negative mean assignment index (m$AI_c$) indicates dispersal whereas positive m$AI_c$ indicates philopatry. Genetic variability ($F_{st}$) was significantly lower in males, and the inbreeding coefficient ($F_{is}$) also indicated male biased dispersal.

## DISCUSSION

Our studies of feral cat ecology have yielded a greater understanding of how cats behave and persist and in cool, dry montane and subalpine environments in Hawai`i. Home range and dispersal rates from population genetics show major differences between sexes, and some male cats had high daily movement rates. Male feral cats on Mauna Kea exhibited some of the largest reported home ranges in the literature and population genetics showed great dispersal distances, perhaps to search for mates. Territorial defense in males may also be an important aspect of behavior. One of our 7 collared males, a subadult, abruptly shifted its home range from west Mauna Kea to the south. Other evidence consistent with aggressive behavior comes from FIV prevalence, which was restricted to males, and is transmitted by biting and scratching during fights (Yamamoto et al. 1988).

Diet analyses showed that invertebrates occur frequently in the diet of feral cats in Hawai`i. This agrees with other Pacific region studies. Dickman (1996), found that invertebrates featured much more prominently in the diets of cats from Pacific region islands (n = 12 studies) than from a range of habitats in mainland Australia (n = 22 studies). Orthoptera were among the most common invertebrate prey of feral cats in the Pacific region. On Mauna Kea, birds occurred more frequently than other major prey types, and rats (*Rattus* spp.) were uncommon prey for cats in both of these subalpine environments. Mice were another important food source for cats on Mauna Loa. Abundant small prey may allow feral cats to survive food shortages, maintain populations at higher densities than they would otherwise be able to maintain, and exploit seasonally abundant prey such as nesting birds.

Our hypothetical model of population dynamics on Hawai`i Island is that abundant birds, including nine introduced gamebird species on Mauna Kea, provide plentiful food resources necessary for high reproduction and survival, thereby creating a potential source population. In such prime habitats, male cats may challenge each other for access to mates, and unsuccessful challengers may be forced to disperse to other marginal locations such as Mauna Loa where bird prey is not as abundant (Scott et al. 1986), or to warmer lowland areas such as Kīlauea. Although low compared to estimates of isolated feral cat populations in urban France ($f$ = 0.14; Say et al. 2003) and on a sub-Antarctic island ($f$ > 0.11; Pontier et al. 2005), we found some evidence of inbreeding in the Mauna Kea populations. Inbreeding may be caused by factors

that cause low population density, such as kitten mortality due to feline leukemia virus, feline immunodeficiency virus, and toxoplasmosis, all of which have been documented in feral cats on Mauna Kea (Danner et al. 2007). Apparently, feral cats in Hawaii still exhibit colonizing characteristics. Although we have no definitive evidence to rule out the possibility that domestic house cats were recruited into nearby feral populations, all of the cats we captured lacked diversity in coat coloration and had reverted back to pelage characteristics similar to European wildcats. Mauna Kea may be considered a source population that can be targeted for control. However, recolonization seems likely given the cats' ability to disperse despite barriers such as lava flows.

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## LITERATURE CITED

ASPI, J., W. ROININEN, M. RUOKONEN, I. KOJOLA, AND C. VILÀ. 2006. Genetic diversity, population structure, effective population size and demographic history of the Finnish wolf population. Molecular Ecology 15:1561–1576.

BANKO, P. C. 1992. Constraints on productivity of wild nene or Hawaiian geese *Branta sandvicensis*. Wildfowl 43:99-106.

BRACKENRIDGE, W. D. 1841. Journal kept while on the U.S. exploring expedition, 1838–1841. Unpublished manuscript at the Maryland Historical Society.

DANNER, R. M, D. M. GOLTZ, S. C. HESS, P. C. BANKO. 2007. Evidence of feline immunodeficiency virus, feline leukemia virus, and *Toxoplasma gondii* in feral cats on Mauna Kea, Hawaii. Journal of Wildlife Diseases 43:315-318.

DICKMAN, C. R. 1996. Overview of the impacts of feral cats on Australian native fauna. Australian Nature Conservation Agency, Canberra, Australia.

EDWARDS, G. P., N. DE PREU, B. J. SHAKESHAFT, I. V. CREALY, AND R. M. PALTRIDGE. 2000. Home range and movements of male feral cats (*Felis catus*) in a semiarid woodland environment in central Australia. Austral Ecology 26:93-101.

ESRI. 1999. ArcView GIS version 3.2. Environmental Systems Research Institute, Redlands, California, USA.

HANSEN, H., S. C. HESS, D. COLE, AND P. C. BANKO. In press. Using population genetic tools to assess control strategies for feral cats (*Felis catus*) in Hawaii. Wildlife Research.

HESS, S. C., P. C. BANKO, D. M. GOLTZ, R. M. DANNER, AND K. W. BRINCK. 2004. Strategies for reducing feral cat threats to endangered Hawaiian birds. Proceedings of the Vertebrate Pest Conference 21:21-26.

HESS, S. C., H. HANSEN, D. NELSON, R. SWIFT, AND P.C. BANKO. In Press. Diet of feral cats in Hawai`i Volcanoes National Park. Pacific Conservation Biology.

HONNOLD, S. P., BRAUN, R., SCOTT, D. P., SREEKUMAR, C., AND DUBEY, J. P. 2005. Toxoplasmosis in a Hawaiian monk seal (*Monachus schauinslandi*). Journal of Parasitology 91:695-697.

HOOGE, P. N., AND B. EICHENLAUB. 1997. Animal movement extension to ArcView. Version 1.1. Alaska Science Center, U.S. Geological Survey, Anchorage, Alaska, USA.

HU, D., C. GILDDEN, J. S. LIPPERT, L. SCHNELL. J. S. MACIVOR, AND J. MEISLER. 2001. Habitat use and limiting factors in a population of Hawaiian dark-rumped petrels on Mauna Loa, Hawai`i. Studies in Avian Biology 22:234-242.

KING, C. 1984. Immigrant killers: introduced predators and the conservation of birds in New Zealand. Oxford University Press, Auckland, New Zealand.

NATIVIDAD-HODGES, C. S., AND R. J. NAGATA. 2001. Effects of predator control on the survival and breeding success of the endangered Hawaiian dark-rumped petrel. Studies in Avian Biology 22:308-318.

PERKINS, R. C. L. 1903. Vertebrata. Pages 365-466 *in* Fauna Hawaiiensis. Sharp, editor. The University Press, Cambridge, United Kingdom.

PONTIER, D., L. SAY, S. DEVILLARD, AND F. BONHOMME. 2005. Genetic structure of the feral cat (*Felis catus*) introduced 50 years ago to a sub-Antarctic island. Polar Biology 28:268-275.

ROTHSCHILD, W. 1893. The avifauna of laysan and the neighboring islands: with a complete history to date of birds of the Hawaiian possessions. R. H. Porter, London.

SAY, L., F. BONHOMME, E. DESMARAIS, AND D. PONTIER. 2003. Microspatial genetic heterogeneity and gene flow in stray cats (*Felis catus*): a comparison of coat colour and microsatellite loci. Molecular Ecology 12:1669-1674.

SCOTT, J. M., S. MOUNTAINSPRING, F. L. RAMSEY, AND C. B. KEPLER. 1986. Forest bird communities of the

Hawaiian islands: their dynamics, ecology, and conservation. Studies in Avian Biology 9.

SEAMAN, D. E., AND R. A. POWELL, 1996. An evaluation of the accuracy of kernel density estimators for home range analysis. Ecology 77:2075-2085.

SIMONS, T. R. 1983. Biology and conservation of the endangered Hawaiian dark-rumped petrel (Pterodroma phaeopygia sandwichensis). National Park Service, Cooperative Studies Unit, University of Washington, CPSU/UW 83-2, Seattle, Washington, D.C., USA.

SMITH, D. G., POLHEMUS, J. T., AND VANDERWERF, E. A. 2002. Comparison of managed and unmanaged wedge-tailed shearwater colonies on O`ahu: effects of predation. Pacific Science 56:451-457.

SWIHART, R. K, AND N. A. SLADE, 1985. Testing for independence of observations in animal movements. Ecology 66:1176-1184.

SWIHART. R. K. AND N. A. SLADE, 1986. The importance of statistical power when testing for independence of animal movements. Ecology 67:255-258.

TOMICH, P. Q. 1986. Mammals in Hawaii. Bishop Museum Press, Honolulu, Hawaii, USA.

VEITCH, C. R. 1985. Methods of eradicating feral cats from offshore islands in New Zealand. Pages 125-155 *in* Conservation of island birds. P. J. Moors, editor. International Council Bird Preservation, Cambridge, England.

WINTER, L. 2003. Popoki and Hawai`i's native birds. `Elepaio 63:43-46.

WORK, T. M., MASSEY, J. G., LINDSAY, D. S., AND DUBEY, J. P. 2002. Toxoplasmosis in three species of native and introduced Hawaiian birds. Journal of Parasitology 88:1040-1042.

WORK, T. M., MASSEY, J. G., RIDEOUT, B. A., GARDINER, C. H., LEDIG, D. B., KWOK, O. C. H., AND DUBEY, J. P. 2000. Fatal toxoplasmosis in free-ranging endangered `Alala from Hawaii. Journal of Wildlife Diseases 36:205-212.

YAMAMOTO, J. K., E. SPARGER, E. W. HO, P. R. ANDERSEN, T. P. O'CONNOR, C. P. MANDELL, L. LOWENSTINE, R. MUNN, AND N. C. PEDERSEN. 1988. Pathogenesis of experimentally induced feline immunodeficiency virus infection in cats. American Journal of Veterinary Research 49:1246–1258.

# DEVELOPING A NEW TOXIN FOR POTENTIAL CONTROL OF FERAL CATS, STOATS, AND WILD DOGS IN NEW ZEALAND

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***Abstract:*** The endemic fauna of New Zealand evolved in the absence of mammalian predators and their introduction has been responsible for many extinctions and declines. Predator control will have to be on-going if some native species are to survive on the mainland. Currently, predator control relies largely on labour-intensive trapping, so the development of humane predator-specific toxins would provide valuable additional control methods. Para-aminopropiophenone (PAPP) is being investigated as a toxin for feral cats (*Felis catus)*, stoats (*Mustela erminea)*, and wild dogs (*Canis familiaris*). Carnivores appear to be much more susceptible to PAPP than birds, so it potentially has a high target specificity, at least in the New Zealand context. Pen trials with 20 feral cats, 15 stoats, and 14 dogs have been undertaken using meat baits containing a proprietary formulation of PAPP. A PAPP dose of 20-34 mg/kg was lethal for feral cats, 37-95 mg/kg was lethal for stoats, and 26-43 mg/kg was lethal for dogs. Our results suggest that PAPP is a humane and effective toxin for control of feral cats and stoats, and possibly for wild dogs. We are now continuing studies towards product registration, which will include the assessment of non-target effects, particularly on birds.

***Key Words:*** *Canis familiaris*, control, *Felis catus*, feral cat, invasive species, *Mustela erminea*, PAPP, para-aminopropiophenone, stoat, toxin, wild dog.

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## INTRODUCTION

New Zealand wildlife evolved in the absence of mammalian predators, and their introduction has been responsible for many extinctions and declines (Clout and Saunders 1995, Parkes and Murphy 2003). Birds have been particularly affected, with over 40% of the pre-human land bird species now extinct, and the proportion of extant birds classed as threatened is one of the highest in the world (Clout 1997). Predator control in New Zealand currently relies largely on labour-intensive trapping (Gillies et al. 2003, Christie et al. 2004), so the development of a humane predator-specific toxin would be a valuable additional control method.

The United States Fish and Wildlife Service investigated para-aminopropiophenone (PAPP) as a toxin for coyote (*Canis latrans*) control in the 1980s as it was highly toxic to canids (Savarie et al. 1983). Vomiting was a complicating factor, but before solutions could be fully explored, the continued use of 1080 and other toxins was approved in the United States (P. Savarie, personal communication). PAPP is currently being investigated in both New Zealand and Australia for the humane control of introduced predators (Marks et al. 2004, Fisher et al. 2005, Murphy et al. 2005, Fleming et al. 2006, Fisher and O'Connor 2007)

The toxic effects of PAPP appear to be related to the rapid formation of methemoglobin in some species (Vandenbelt et al. 1944). High methemoglobin levels reduce the oxygen-carrying capacity of the blood and can result in death by hypoxia. Carnivore species appear to be much more susceptible than birds (Savarie et al. 1983, Schafer et al. 1983), so PAPP potentially has some degree of target specificity in the New Zealand situation at least.

A proprietary formulation of PAPP has been developed by Connovation Ltd., New Zealand. It delivers PAPP rapidly into the blood stream of an animal via the stomach and can be used in a meat bait. The aim of this study was to evaluate the effectiveness of this formulation on dogs, cats, and stoats.

## METHODS

PAPP was milled to produce a consistent particle size, mixed with carriers, and made into pellets or microgranules to be mixed or placed with

food. Each dose contained a propriety formulation of PAPP which included an anti-emetic.

Ten dogs were fed chicken (*Gallus domesticus*) mince containing 310-1,200 mg PAPP and 4 dogs were fed dog roll containing a 600 mg capsule of PAPP. The dogs were not fasted beforehand and all ate the bait.

Twenty individually-housed cats were fasted for 24 hours and then presented with 10-20 g of feed containing 80-180 mg PAPP. The cats were tested in two groups. In the first group, 12 cats weighing 0.9-2.6 kg received doses of 35-112 mg/kg. In the second group, 8 cats weighing 2.4-3.9 kg received doses of 20-34 mg/kg.

Fifteen individually-housed stoats weighing 135-345 g were fasted for 24 hours and then presented with 10 g of minced rabbit (*Oryctolagus cuniculus*) containing 12.8 mg of PAPP in a paste.

All animals had free access to water and were observed for signs of toxicosis after dosing. To determine if there was a significant relationship between the dose (in the range used) and time to death, a regression analysis was done by log-transforming both dose and time to death.

## RESULTS

Ten of 14 dogs (weight range 14-30 kg) fed 21.4-42.9 mg/kg of PAPP in meat baits died within 3.5 hours of bait ingestion (Table 1). There was no significant relationship between the dose (mg/kg) and time to death ($F [1,8] = 0.59$, P [one-tailed] = 0.23). The four dogs that survived received 11.1-25.8 mg/kg of PAPP and all vomited.

Eighteen of the 20 cats (weight range 0.9-3.9 kg) died after eating a PAPP bait and two survived (Table 2). In the higher dose group, the twelve cats showed a wide range of responses with onset of symptoms occurring between 16-204 min and death between 37-246 min. One animal that only ate a fraction of the dose of PAPP recovered (see Table 2, first cat). In the lower dose group, onset of symptoms occurred between 22-55 min with death between 54-125 min. Cats exposed to lethal doses of PAPP tended to lose consciousness without spasms or convulsions. There was no significant relationship between the dose (mg/kg) and time to death ($F [1,16] = 0.19$, P [one-tailed] = 0.34).

All 15 stoats died quickly after eating PAPP baits (Table 3). For stoats (weight range 135-345 g, dose 37.1-94.8 mg/kg), 12.8 mg of PAPP was lethal in all cases, with first symptoms 6-40 min after ingestion and death between 15-85 min. There were no signs of discomfort, stress, or vomiting associated with poisoning. The stoats became quiet, lethargic, and then unconscious for a short period before death. There was a significant relationship between the dose (mg/kg) and time to death ($F [1,13] = 3.8096$, P [one-tailed] = 0.036).

## DISCUSSION

The lethal response of dogs, cats and stoats to PAPP was rapid and appeared relatively free of the suffering that accompanies the use of some other toxins (Goh et al. 2005, Potter at al. 2006). PAPP was effective when delivered in a range of different

**Table 1.** Time to first symptoms, vomit, and death of 14 dogs presented with 310 to 1,200 mg PAPP placed into a meat bait. NA = did not vomit. S = survived.

| Weight (kg) | Dose (mg/kg) | First symptoms (min) | Time to vomit (min) | Time to death (min) | Bait |
|---|---|---|---|---|---|
| 28 | 11.1 | 140 | 158 | S | Chicken |
| 20 | 15.8 | 70 | 83 | S | Chicken |
| 18 | 17.5 | 110 | 120 | S | Chicken |
| 28 | 21.4 | 70 | 70 | 114 | Dog roll |
| 24 | 25.0 | 97 | 140 | 191 | Dog roll |
| 12 | 25.8 | 128 | 135 | S | Chicken |
| 21 | 28.6 | 62 | NA | 104 | Dog roll |
| 20 | 30.0 | 34 | 38 | 71 | Chicken |
| 21 | 30.7 | 46 | 55 | 111 | Chicken |
| 19 | 31.6 | 112 | NA | 178 | Chicken |
| 16 | 37.5 | 94 | NA | 152 | Chicken |
| 30 | 40.7 | 80 | 91 | 151 | Chicken |
| 15 | 41.4 | 114 | NA | 210 | Dog roll |
| 14 | 42.9 | 45 | NA | 118 | Chicken |

**Table 2.** Time to first symptoms and death of 20 cats presented with 10-20 g of bait containing 80 to 180 mg PAPP. S = survived. Rabbit = minced rabbit.

| Weight (kg) | Dose (mg/kg) | First symptoms (min) | Time to death (min) | Bait |
|---|---|---|---|---|
| 0.9 | ? | 26 | S | Did not eat |
| 2.6 | 34.6 | 45 | 119 | Dog roll |
| 2.4 | 37.8 | 54 | 73 | Dog roll |
| 2.4 | 37.8 | 60 | 101 | Dog roll |
| 2.5 | 40.2 | 45 | 77 | Dog roll |
| 1.7 | 53.3 | 204 | 246 | Beef fat |
| 1.6 | 54.9 | 16 | 37 | Dog roll |
| 1.3 | 67.2 | 26 | 74 | Did not eat |
| 1.2 | 76.3 | 136 | 224 | Beef fat |
| 1.1 | 80.4 | 66 | 86 | Peanut butter |
| 0.9 | 96.8 | 78 | 108 | Peanut butter |
| 1.6 | 111.8 | 25 | 55 | Dog roll |
| 3.9 | 20.4 | 35 | 92 | Rabbit |
| 3.5 | 22.7 | 31 | 67 | Rabbit |
| 3.4 | 23.7 | 55 | 125 | Rabbit |
| 3.3 | 24.3 | 35 | 74 | Rabbit |
| 3.1 | 25.6 | 22 | 54 | Rabbit |
| 2.5 | 32.0 | 30 | 83 | Rabbit |
| 2.8 | 32.4 | 50 | S | Rabbit |
| 2.4 | 33.9 | 30 | 79 | Rabbit |

**Table 3.** Time to first symptoms and death of 15 stoats presented with 10 g minced rabbit containing 12.8 mg PAPP in a paste.

| Weight (g) | Dose (mg/kg) | First symptoms (min) | Time to death (min) |
|---|---|---|---|
| 345 | 37.1 | 15 | 45 |
| 321 | 39.9 | 10 | 85 |
| 298 | 43.0 | 17 | 43 |
| 295 | 43.4 | 15 | 35 |
| 292 | 43.8 | 15 | 62 |
| 281 | 45.6 | 7 | 35 |
| 249 | 51.4 | 20 | 55 |
| 232 | 55.2 | 10 | 20 |
| 210 | 61.0 | 20 | 59 |
| 199 | 64.3 | 15 | 35 |
| 198 | 64.6 | 35 | 45 |
| 192 | 66.7 | 20 | 51 |
| 185 | 69.2 | 15 | 30 |
| 163 | 78.5 | 40 | 51 |
| 135 | 94.8 | 6 | 15 |

vehicles or baits, such as minced rabbit, beef fat, and dog roll. The two cats fed the PAPP in beef fat had the longest time till first symptoms and longest time to death, suggesting that fat may slow the absorption of PAPP.

In stoats, the symptoms of poisoning and time to death after ingestion of baits were similar to those previously reported by Fisher et al. (2005) following administration of PAPP in the hydrochloride form by oral gavage.

A previous complication of using PAPP was that some animals vomited after ingesting a bait, thereby reducing its effectiveness (Savarie et al. 1983, Murphy et al. 2005, Fisher and O'Connor 2007). This was not a problem with cats and stoats in this study. However, vomiting did occur in 9 of 14 dogs and all four that survived had vomited. The unusually large weight range observed in dogs (Vila 1999) may also be a complicating factor in the field use of PAPP (and other toxins).

Our results suggest that PAPP is a humane and effective toxin for control of feral cats and stoats (and possibly wild dogs) and we are now continuing studies towards product registration. The registration requirements will have to be met of both the Environmental Risk Management Authority (ERMA) - who administers the Hazardous Substances and New Organisms (HSNO) Act 1996 - and the NZ Food Safety Authority under the Agricultural Chemicals and Veterinary Medicines (ACVM) Act 1997. These studies will include the assessment of non-target effects, particularly on birds. Although birds are not as susceptible to PAPP as carnivores on a weight to weight basis, because they generally have lower mass, they could still be vulnerable. For example, the red-winged blackbird (*Agelaius phoeniceus*) has an $LD_{50}$ of 133 mg/kg (Savarie et al. 1983) which equates to only 9 mg of PAPP. This is slightly less than a lethal dose for stoats, but considerably less than a lethal dose for cats. A list of New Zealand birds that potentially could be vulnerable (depending on their $LD_{50}$) to PAPP stoat or cat baits was presented by Murphy et al. (2005). It may be possible to improve specificity by developing more targeted delivery systems if required (e.g., Marks et al. 2004, Marks et al. 2006).

A PAPP bait could be particularly useful for stoats, because currently there are no toxins registered for use against them. Stoats continue to have an impact on a wide range of threatened birds, lizards and invertebrates in New Zealand, and there are few effective techniques available to control them (Parkes and Murphy 2004). Registration of a toxin that appears to show some relative specificity for mammals and is lethal to stoats at low dose would be a significant advance.

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## LITERATURE CITED

CHRISTIE, J. E., E. C. MURPHY, AND I. WESTBROOKE. 2004. An inventory of stoat trapping operations in 2001/02 and an assessment of the data collection methods. DOC Science Internal Series 177. Department of Conservation, Wellington, New Zealand.

CLOUT, M. 1997. Predator management in New Zealand: an overview. Pages 3-5 *in* J. Sim, and A. J. Saunders, editors. National Predator Management Workshop 1997. Department of Conservation, Wellington, New Zealand.

CLOUT, M., AND A. J. SAUNDERS. 1995. Conservation and ecological restoration in New Zealand. Pacific Conservation Biology 2:91-98.

FISHER, P. M., C. E. O'CONNOR, AND E. C. MURPHY. 2005. Acute oral toxicity of p-aminopropiophenone to stoats. New Zealand Journal of Zoology 32:163-169

FISHER, P., AND C. O'CONNOR. 2007. Oral toxicity of p-aminopropiophenone to ferrets. Wildlife Research 34:19-24.

FLEMING, P. J. S., L. R. ALLEN, S. J. LAPIDGE, A. ROBLEY, G. R. SAUNDERS, AND P. C. THOMSON. 2006. A strategic approach to mitigating the impacts of wild canids: proposed activities of the Invasive Animals Cooperative Research Centre. Australian Journal of Experimental Agriculture 46:753-762.

GILLIES, C. A., M. R. LEACH, N. B. COAD, S. W. THEOBALD, J. CAMPBELL, T. HERBERT, P. J. GRAHAM, AND R. J. PIERCE. 2003. Six years of intensive pest mammal control at Trounson Kauri Park, a Department of Conservation "mainland island", June 1996-July 2002. New Zealand Journal of Zoology 30:399-420.

GOH C. S., D. R. HODGSON, S. M. FEARNSIDE, J. HELLER, AND N. MALIKIDES. 2005. Sodium monofluoroacetate (Compound 1080) poisoning in dogs. Australian Veterinary Journal 83:474-479.

MARKS, C. A., F. GIGLIOTTI, F. BUSANA, M. JOHNSTON, AND M. LINDEMAN. 2004. Fox control using a para-aminopropiophenone formulation with the M-44 ejector. Animal Welfare 13:401-407.

MARKS, C. A., M. J. JOHNSTON, P. M. FISHER, K. PONTIN, AND M. J. SHAW. 2006. Differential particle size ingestion: promoting target-specific baiting of feral cats. Journal of Wildlife Management 70:1119-1124.

MURPHY, E. C., A. LAVRENT, D. MACMORRAN, L. ROBBINS, AND P. ROSS. 2005. Development of a humane toxin for the control of introduced mammalian predators in New Zealand. Proceedings of the Australasian Vertebrate Pest Conference 13:137-142.

PARKES, J., AND E. C. MURPHY. 2003. Management of introduced mammals in New Zealand. New Zealand Journal of Zoology 30:335-359.

PARKES, J., AND E. C. MURPHY. 2004. Risk assessment of stoat control methods for New Zealand. Science for Conservation 237.

POTTER, M. A., D. P. BARRETT, AND C. M. KING. 2006. Acceptance by stoats (*Mustela erminea*) of 1080 (sodium monofluroacetate) in small-volume baits and its effect on behaviour and time to death. New Zealand Veterinary Journal 54:350-356.

SAVARIE, P. J., H. PING PAN, D. J. HAYES, J. D. ROBERTS, G. L. DASCH, R. FELTON, AND E. W. SCHAFER JR. 1983. Comparative acute oral toxicity of *para*-aminopropiophenone. Bulletin of Environmental Contamination and Toxicology 30:122-126.

SCHAFER, E. W., W. A. BOWLES, AND J. HURLBUT. 1983 The acute oral toxicity, repellancy and hazard potential of 998 chemicals to one or more species of wild and domestic birds. Archives Environmental Contamination and Toxicology 12:355-382.

VANDENBELT, J. M., C. PFEIFFER, M. KAISER, AND M. SIBERT. 1944. Methemoglobinemia after administration of p-aminoacetophenone and p-aminopropiophenone. The Journal of Pharmacology and Experimental Therapeutics 80:31-38.

VILA, C. 1999. Phylogenetic relationships, evolution, and genetic diversity of the domestic dog. Journal of Hereditry 90:71-77.

# DEVELOPING A DECISION SUPPORT SYSTEM FOR IMPROVING POSSUM CONTROL PLANNING

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***Abstract:*** Brushtail possums are a major invasive pest in New Zealand. A decision support system (DSS) has been developed to enable those involved in operational planning or delivery of possum control to access available relevant knowledge when making operational decisions. The DSS comprises checklists that ensure users consider relevant constraints and issues, a database of best-practice information on possum control, and an "expert" system that recommends actions based on information provided by the user.

***Key Words:*** decision support system, invasive species, possum, management, *Trichosurus vulpecula*, vertebrate pest.

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## INTRODUCTION

The brushtail possum (*Trichosurus vulpecula*) was introduced into New Zealand from Australia in the mid- to late-1800s, and has become a widespread pest. In particular, the possum is a significant wildlife host and vector of bovine tuberculosis (Tb, *Mycobacterium bovis*), and about NZ$50 million is spent annually on controlling possum numbers in disease-risk areas. Control is also carried out over about 1.1 million ha to limit their damage to conservation areas. Possum control has been supported over several decades by research focused on improving control tools and understanding possum behaviour and ecology (Montague 2000). Consequently, there is a wealth of information available to possum control managers and contractors who undertake possum control on the relative effectiveness and cost of various control tools, control strategies, non-target and environmental impacts, possum biology and behaviour, and legal and biological constraints to operational procedures. Unfortunately, much of this information is found only in formally published scientific papers and books or in unpublished reports, and as a consequence is not readily available to many of the relevant "end-users".

## DEVELOPMENT OF A DECISION SUPPORT SYSTEM

A decision support system (DSS) was developed to enable those involved in operational planning or delivery of possum control to access available relevant knowledge when making operational decisions. DSSs have been developed primarily for the agriculture industry to improve the transfer of information from researchers to end-users and to help the latter interpret and apply it to a range in agricultural systems, including invertebrate pests (Stuth and Smith 1993, El-Azhary et al. 2000, Ghosh and Samanta 2003), grazing systems (Sinclair and Rickert 2000), and diagnoses of animal diseases (Kroschewski et al. 2006, Eisler et al. 2007).

The Possum DSS has three components: (1) Checklists that ensure users consider relevant constraints and issues when planning a possum control operation. Two checklists containing a range of constraints were developed, one for vector managers and one for control contractors. For disease vector managers, the constraints identified include vector control boundaries, stratification of the proposed control area, existing possum density, strategic control options, legislation underpinning control, and post-control possum population monitoring and auditing. For possum control contractors, the constraints include their capability to do the job using the techniques chosen,

technique cost-effectiveness, relevant legislation, environmental risks arising from control (including risks to non-target species), public opposition, and the behavioural responses of possums to the control devices planned for use. (2) A database of best-practice information on possum control. The information is presented as a series of points rather than as large blocks of text in order to make the information more readable. Additional "supplementary" information was collated and linked using hyperlinks through keywords to allow the user to easily locate other information as required. All information is accessible, either directly through a "best-practice index" or through the links provided with the recommended actions, and includes information covering both the mechanics of control (e.g. the specifications of carrot and cereal bait and protocols for aerial delivery, baits for ground control, use of bait stations, bait shyness, pre-feeding, and use of trap and trap suppliers). Also included are strategies for control (e.g., what approaches to use in areas receiving high and low rainfall, areas controlled within the previous three years, areas containing Tb-infected deer and pigs [including how to handle infected carcasses], areas containing game animals considered to be important hunting resources, and areas containing protected wildlife). Additional issues related to possum control such as human health and environmental contamination are also documented. (3) An "expert" system that recommends actions based on information provided by the user. Twenty-nine key rules were identified using 10 control scenarios developed by the authors. A set of 24 queries (prerequisites) was developed that users of the system must consider and identify as true or false. These prerequisites are then submitted to an inference engine, which provides the system interrogator with a set of recommended actions. Mandarax software was selected as the DSS because (1) it is open-source software (i.e. freely available) licensed under the non-invasive LGPL (Lesser General Public Licence) system, (2) expertise in using the system was available in New Zealand, and (3) it has components that make it easy to integrate into web-based systems. Mandarax uses backward-chaining inference (reasoning), and the conclusions of the rules are the actions recommended by the system. This system enables the end-users to better understand the logic used by the system, which increases the trust they have for the system. This in turn encourages them to provide feedback to system managers on the rules used. For the latter, the system facilitates feedback via email. This function turns the system into a two-way communication and knowledge-sharing tool.

A draft DSS (http://possumdss.landcareresearch.co.nz) was developed and made publicly accessible on the Web in early 2006.

## ACKNOWLEDGMENTS

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## LITERATURE CITED

EISLER, M. C., J. W. MAGONA, N. N. JONSSON, AND C. W. REVIE. 2007. A low cost decision support tool for the diagnosis of endemic bovine infectious diseases in the mixed crop-livestock production system in sub-Saharan Africa. Epidemiology and Infection 135:67-75.

EL-AZHARY, E., H. A. HASSAN, AND A. RAFEEA. 2000. Pest control expert system for tomato (PCEST). Knowledge and Information Systems 2:242-257.

GHOSH, I., AND R. K. SAMANTA. 2003. TEAPEST: an expert system for insect pest management in tea. Applied Engineering in Agriculture 19:619-625.

KROSCHEWSKI, K., M. KRAMER, A. MICKLICH, C. STAUBACH, R. CARMANNS, AND F. J. CONRATHS. 2006. Animal disease outbreak control: the use of crisis management tools. Revue scientifique et technique de l'Office International des Epizooties 25:211-221.

MONTAUGE, T. L. 2000. The brushtail possum: biology, impact and management of an introduced marsupial. Manaaki Whenua Press, Lincoln, New Zealand.

SINCLAIR, S. E., AND K. R. RICKERT. 2000. An overview of the incorporation of management systems for red and rusa deer in Queensland within a decision support system. Asian–Australian Journal of Animal Sciences 13:291-294.

STUTH, J. W., AND M. S. SMITH. 1993. Decision support for grazing lands: an overview. Pages 1-35 *in* Decision support systems for the management of grazing lands. J. W. Stuth and B. G. Lyons, editors. UNESCO and The Parthenon Publishing Group: Paris, France.

# THE POLITICAL, ECONOMIC AND MANAGEMENT HISTORY OF A SUCCESSFUL EXOTIC ERADICATION: THE CASE OF BLACK-TAILED JACKRABBITS IN ONE PART OF FLORIDA

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***Abstract:*** Black-tailed jackrabbits are a prolific, mobile species that became established at Miami International Airport, Florida. These animals posed a potential threat for colonization through additional parts of Florida if they were able to spread beyond the expansive airport property. Moreover, their carcasses from collisions with vehicles and aircraft attracted large scavenging and predatory birds to the airport, causing potential airstrike hazards. While the jackrabbits ultimately were successfully eradicated, the political, economic and management paths to that success were convoluted. We describe that history here and how a beneficial outcome was ultimately achieved.

***Key Words:*** airstrike hazard, invasive species, *Lepus californicus*, management economics.

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## INTRODUCTION

Black-tailed jackrabbits (*Lepus californicus*) are not native to Florida, but by 2003 they had been well-established at Miami International Airport (MIA) for many years. How and when they were introduced to this expansive airport property (1307 ha) was unknown. Speculations as to their origins included escapes from a rabbit farm or escapes from transit to dog racing tracks for use in training greyhounds. By 2003, the black-tailed jackrabbit population at MIA was considered to be around 500.

Occupation of the MIA property by a large number of black-tailed jackrabbits posed two potential threats. First, while MIA is relatively encapsulated by the Miami metro area, the jackrabbits still posed a significant invasive threat for other parts of Florida. The species is highly fecund, with 2-7 litters/year and 2-5 young/litter (Flinders and Chapman 2003). Moreover, they are a mobile, fast-moving species (up to 56 kph [Flinders and Chapman 2003]), and once outside the confines of Miami could spread through other parts of Florida (and potentially beyond). The other potential problem their population posed was to cause an increase in bird airstrike hazards. Black-tailed jackrabbits were occasionally killed by collisions with aircraft and vehicles, or the back-blast from jet engines. Their carcasses proved highly attractive to vultures (*Cathartes aura*, *Coragyps atratus* ) for forage. This created a considerable air safety concern, as vultures present significant hazards to aircraft while taking off or landing (e.g., Dolbeer et al. 2000). Besides safety concerns, bird strikes also result in lost revenue and very costly repairs to aircraft (Milsom and Horton 1990, Linnell et al. 1996). With the construction of a new runway, the problem was exacerbated by the decrease in green space the rabbits could occupy and an increase in space with aircraft and vehicular traffic that raised the number of killed rabbits. In fact, removal of the black-tailed jackrabbit population at MIA was instigated as a response to Federal Aviation Administration (FAA) regulations mandating the problem be solved for safety reasons. From March 2001 to March 2003 at least two dozen vultures were struck by aircraft at MIA. We discuss the political aspects, operational methods and their chronologies that eventually led to eradication of black-tailed jackrabbits at MIA before they could further expand their range.

## DEVISING A PLAN

Suggestions for approaching the black-tailed jackrabbit problem were numerous. Miami's Metrozoo called for their extermination, as they believed live-capture was unlikely to eliminate all animals. The FAA concurred and recommended

that the US Department of Agriculture, Wildlife Services (WS), the Federal agency responsible for managing conflicts with wildlife (US Department of Agriculture 1997), carry out the eradication. At the same time, animal rights activists strongly urged live-capture and transfer of the black-tailed jackrabbits to their native range. Additionally, the recommendations from the general public included fencing, shooting, poisoning, sterilization, and sale to dog breeders. At that time, there was an increasing trend in bird airstrikes at MIA. Between 21 April and 21 August 2003, there were nine reported bird airstrikes at MIA. Most airports account for less than 20% of the actual number of strikes (Dolbeer et al. 1995). Wildlife strike statistics based on pilot reports generally are incomplete, because pilots either do not report strikes or the proportion of reported strikes varies due to factors such as decreased pilot acuity towards wildlife during critical phases of flight, size of the animal, group size, weather conditions, time of day, or heightened pilot awareness during migratory seasons (Linnell et al.1999). Thus, it is logical to assume that the same held true for MIA, and that more strikes may have occurred than have been reported.

The initial action plan by the Miami-Dade County Aviation Department (MDCAD), responsible for managing MIA, was to form an agreement with WS for the eradication of the black-tailed jackrabbits. However, once the plans for lethal control became known to the public, there was a vociferous outcry, a media frenzy, and a public relations nightmare. This generated substantial political pressure to forestall the eradication and use non-lethal means to remove and relocate the black-tailed jackrabbits from MIA.

The eradication was temporarily rejected and a modified plan was introduced. This second action plan called for the black-tailed jackrabbits to be live-captured by a reputable private firm and housed for translocation to a sanctuary near Hutchins, Texas, where they are native. Finally, any black-tailed jackrabbits remaining at MIA after a 2-month capture period would be eradicated by various methods by the WS.

## PLAN IMPLEMENTATION AND DÉJÀ VU

Black-tailed jackrabbits were captured by several non-lethal methods including cage traps, hand capture (by hand or net), and tranquilization using dart guns. Nevertheless, a substantial number of black-tailed jackrabbits still remained at MIA when the time period for live-capture expired. This triggered the final stage of the plan, eradication of the remaining jackrabbits by WS. This in turn triggered another public outcry and a fight by animal activists to extend the time period in which the private contractor could live-trap black-tailed jackrabbits. Again, strong political pressure was built and eradication was delayed as the live-capture methods were given another (third) month to succeed. In the meantime, concerns about the safety problems at MIA continued.

Black-tailed jackrabbits remained at MIA at the expiration of the extension of the live-capture period. By this time 301 black-tailed jackrabbits had been captured at MIA (210 by cage traps, 45 by hand capture, 46 by tranquilizer dart) and another 34 inadvertently killed during capture attempts. Only 12-30 jackrabbits were claimed to remain at the end of this period (Rabin 2003a). The eradication portion of the action plan was again triggered. However, a strategy not completely successful on the ground might still succeed in court. Therefore, animal activist groups went to court to attempt to judicially impose more time for live-capture methods to succeed at complete black-tailed jackrabbit removal. The animal activists lost their appeal in court with the ruling citing that continued delays would jeopardize public safety. At this point, black-tailed jackrabbit removal reverted to the original approach by MDCAD: eradication by WS. As this phase moved forward, it was accompanied by protests and dramatic headlines in newspapers. In contrast to the above assessment of the number of jackrabbits remaining, newspapers reported that WS removed 35 on the first night of shooting (Rabin 2003b). In reality, WS removed 44 on the first night, 55 on the second night, and 172, overall, to complete the eradication.

## MANAGEMENT ECONOMICS

The agreement for WS to eradicate the black-tailed jackrabbit at MIA was for 80 hrs of effort and a cost of $19,549. The initial estimate provided for live-capture and translocation of all black-tailed jackrabbit at MIA to Texas was about $7,500 (Associated Press 2003). Ultimately, it cost in excess of $50,000 to live-capture and translocate about 60% (301) of the MIA black-tailed jackrabbit population to Texas. The cost per rabbit for live-capture and translocation was, therefore, at least $166, whereas it cost $114 per rabbit to shoot the remainder. Moreover, as an animal removal

operation proceeds (by whatever methods), the amount of effort required per animal typically increases. The animals removed by WS were the last third of the population to be removed, thereby probably representing the most difficult portion of the population to eliminate. Because the WS contract was a fixed amount for black-tailed jackrabbit removal on airport property, we surmise that the cost per rabbit would have only been $39 had they removed all of the jackrabbits.

## DISCUSSION

Florida joins Hawaii as the two states with the most severe invasive species problems in the United States (U.S. Congress 1993), with breeding populations of introduced or invasive vertebrate species regularly identified. The impacts from many introductions are unknown or not readily perceived by the public, while others are immediately apparent or have their negative potential revealed over time. Even highly prolific invasive species may fester for a considerable time before exhibiting an explosive expansion of their range (Shigesada and Kawasaki 1997). This was the case with the black-tailed jackrabbit at MIA. While their population anecdotally appeared to be in a relatively rapid increase phase, their true negative impacts to Florida's habitats would only have occurred had they dispersed beyond Miami. Fortunately, their population was extirpated before this could take place. Nevertheless, the existence of this invasive species at MIA created a potential threat to human health and safety as well as an economic burden for the damage to aircraft.

Although surveys continue at MIA, the eradication of black-tailed jackrabbits at MIA does not mean vigilance for the species should not be maintained in Florida. One other breeding population of black-tailed jackrabbits (in Broward County) had existed and has been extirpated. They also have been observed in other parts of Florida, but not as breeding populations (Florida Wildlife Commission, unpublished data). Black-tailed jackrabbits could easily be illegally or accidentally released in Florida again. Should that happen, the experiences from the MIA eradication could provide direction on how an effective and efficient removal program could be conducted.

Management of an exotic species requires more than the recognition of a potential problem, it also requires a government and public motivation to address the problem. In the case of the black-tailed jackrabbit, the human safety issue motivated their removal, rather than their potential for ecological harm should they have dispersed from MIA. Had it not been for their exacerbation of airstrike hazards at MIA, it is unlikely they would have been eradicated and their population would have continued to be a festering threat for eventual dispersal.

## LITERATURE CITED


ASSOCIATED PRESS. 2003. Miami airport weighs offer to rid rabbits from grounds. Gainesville Sun, March 30, 2003.

DOLBEER, R. A., S. E. WRIGHT, AND E. C. CLEARY. 1995. Bird and other wildlife strikes to civilian aircraft in the United States, 1994. Interim report DTFA011-91-Z-02004. US Department of Agriculture For Federal Aviation Administration, FAA Technical Center, Atlantic City, New Jersey, USA.

DOLBEER, R. A., S. E. WRIGHT, AND E. C. CLEARY. 2000. Ranking the hazard level of wildlife species to aviation. Wildlife Society Bulletin 28:372-378.

FLINDERS, J. T. AND J. A. CHAPMAN. 2003. Black-tailed jackrabbit. Pages 126-146 *in* G. A. Feldhammer, B. C. Thompson, and J. A. Chapman, editors. Wild mammals of North America: biology, management, and conservation. Johns Hopkins University Press, Baltimore, Maryland, USA.

LINNELL, M. A., M. R. CONOVER, AND T. J. OHASHI. 1996. Analysis of bird strikes at a tropical airport. Journal of Wildlife Management. 60:935-945.

LINNELL, M. A., M. R. CONOVER, AND T. J. OHASHI. 1999. Biases in bird strike statistics based on pilot reports. Journal of Wildlife Management. 63:997-1003.

MILSOM, T. P., AND N. HORTON. 1990. Birdstrike: an assessment of the hazard on UK civil aerodromes from 1976-1990. United Kingdom Bird Strike Avoidance Team.

RABIN, C. 2003a. Animal lover loses bid to save remaining rabbits at airport. Miami Herald, July 11, 2003.

RABIN, C. 2003b. All but a few hares at airport are shot. Miami Herald, July 12, 2003.

SHIGESADA, N. AND K. KAWASAKI. 1997. Biological invasions: theory and practice. Oxford University Press, Oxford, United Kingdom.

U.S. CONGRESS. 1993. Harmful non-indigenous species in the United States. Office of Technology Assessment, OTA-F-565, Washington, D.C., USA.

U.S. DEPARTMENT OF AGRICULTURE/Animal and Plant Health Inspection Service, U.S. Department of Agriculture/Forest Service and Department of Interior/Bureau of Land Management. 1997. Animal damage control program final environmental impact statement (revised). USDA Animal and Plant Health Inspection Service. Washington, D.C., USA.

# VERTEBRATE INVASIONS IN CHILE: AN INVITATION TO COLLABORATION

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***Abstract:*** Chile is an interesting country for the study of biological invasions because it is flanked by biogeographical barriers on all sides. Therefore, it should be relatively immune from invasions. Indeed, the incidence of invasive species in regions comparable to Chile (California, mediterranean Europe, South Africa, mediterranean Australia) is generally higher, except for mammals. The most likely cause of the high incidence of invading mammals is their previous introduction to Argentina and subsequent spread from there toward Chile, crossing the Andes Ranges. Thus, this supposed barrier to immigration is ineffective. I analyze four invasive species that Chile shares with the United States (US) and/or Canada: African clawed frog (*Xenopus laevis*), California quail (*Callipepla californica*), American beaver (*Castor canadensis*), and European rabbit (*Oryctolagus cuniculus*). I introduce three main topics on which collaborative work could be initiated between US and Chilean researchers: (1) invasion pathways, spread rates, and environmental impact of shared species, (2) comparative ecology of shared species, and (3) genetic diversity of populations of shared species.

***Key Words:*** biogeographical barriers, California, Canada, Chile, collaborative work, invasive species, mediterranean-type regions, shared species, species translocations, United States.

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## AN ISOLATED COUNTRY THAT IS NOT

Chile is an interesting country for the study of biological invasions: it is not only long and narrow but is flanked by strong biogeographically barriers on all sides (Atacama desert to the north, Andean Ranges to the east, Pacific Ocean to the west) and is separated from other southern continents by broad oceans, the Pacific and the Atlantic (Jaksic 1998, Figueroa et al. 2004). Nevertheless, Chile currently hosts at least one invasive species of each amphibians and reptiles (2.4% and 1.1% of the respective native species), seven of birds (1.8% of the total number of native bird species), and 15 invasive mammals (accounting for 15.2% of the respective native species). Thus, this assumedly isolated country is not immune from being invaded.

The raw numbers and percentages quoted above do not seem too impressive, but may be misleading. It makes more sense to compare the incidence of invasive birds and mammals in regions climatically and vegetationally similar to Chile. For instance, California, southern Europe, South Africa, and southern Australia, the so-called mediterranean regions of the world (Jaksic 1998, Figueroa et al. 2004). In comparison to these similar regions, Chile has relatively few invasive birds (2.7% in mediterranean Chile versus 3.5% in mediterranean Europe, and 6.7% in mediterranean California). But it does have more invasive mammals (7.5% in mediterranean Chile versus 5.6% in mediterranean California). The most likely cause of the high incidence of invading mammals in Chile is their previous introduction to Argentina and subsequent spread from there toward Chile, crossing the Andes Ranges (Jaksic et al. 2002, Iriarte et al. 2005). This conveys the lesson that even a very tall mountain range is not barrier enough for invading mammals.

All of this general information has been published in previous papers cited above. Chile, Canada, and the US share several species in common, among them, monk parakeet (*Myiopsitta monachus*), muskrat (*Ondatra zibethicus*), nutria (*Myocastor coypus*), feral pig (*Sus scrofa*), and American mink (*Mustela vison*). Here, I will refer to other four species that Chile shares with the United States (US) and Canada as invaders in that country, with the aim of eliciting interest in collaboration between the respective country scientists.

## FOUR INVADING VERTEBRATES IN CHILE

### African clawed frog, *Xenopus laevis*

There is only one amphibian invader in Chile, the African clawed frog. This species was introduced to a lagoon close to the international airport of Santiago in 1973, and then spread out in all directions except towards the Andes mountains (Jaksic 1998, Lobos and Jaksic 2005). This invasion wave has traveled over 500 km in 34 years which is equal to 15 km/year! But this is an overestimate, because there have been many intentional translocations. African clawed frogs reach from almost sea level up to 620 m and are present more often in lagoons (94% of sites) than in rivers (60% of sites). With regard to lagoons, they are present mostly in irrigation dams (80% of occurrences) than in natural lagoons (20%). African clawed frogs inhabit quite a diverse array of habitats with regards to water temperature, dissolved oxygen, pH, and electric conductivity, indicating a high degree of adaptability and further invasion potential. In central Chile, African clawed frogs prey on essentially three major food types: insects, mollusks and crustaceans, with *Xenopus* larvae being the only vertebrates found in local diets. It has been hypothesized that the African clawed frog is little preyed upon because of its toxic skin. But we have quantitative records that at least three Chilean native bird species prey on clawed frogs with the main predator being night heron (*Nycticorax nycticorax*), but also kelp gull (*Larus dominicanus*) and burrowing owl (*Speotyto cunicularia*). The African clawed frog is considered to be completely aquatic at all stages of its life history, but we have quantitative records of overland migration during nighttime, which increases its invasive potential.

### California quail, *Callipepla californica*

The exact distributional range of California quail in Chile is unknown, but it is found from at least Antofagasta (Chile's II Region) to Puerto Montt (X Region) and also in oceanic Robinson Crusoe Island (offshore from the V Region). The history of introductions and translocations of this species is quite interesting (Jaksic 1998, Swarth 1927). It appears to have been introduced from California to at least two different parts of Chile on different occasions during the 1860s and 1880s. According to Swarth (1927), William Groves brought a dozen quail from San Jose, California, to Limache (Chile's V Region) in 1864, and kept them caged until all escaped, establishing a population in the wild. Later, C. J. Lambert imported quail from San Francisco, California, and released them to the wild in La Serena (Chile's IV Region) in 1881 or 1882. In an unexpected twist, quail were reintroduced from Chile back to California in the 1920s. Indeed, F. E. Booth bought quail in Santiago (Metropolitan Region) and shipped them back to San Francisco, where a first batch of 134 quail arrived in January 1927 and was sent to a game reserve in Sonoma county. The paper reporting all this information (Swarth 1927) was submitted for publication on February 9, 1927, and thus, it is possible that other "Chilean" quail may have been shipped back to California! Perhaps this could be traced using genetic markers.

### American beaver, *Castor canadensis*

The American beaver was first introduced from around the Toronto area of Canada to the Argentinean side of Tierra del Fuego Island in 1946 (Jaksic 1998, Jaksic et al. 2002, Iriarte et al. 2005, Wallem et al., In Press). From their release site, American beaver expanded toward the Chilean half of the island and to the rest of the Fuegian archipelago. This species was thought to be confined to the south of the Straits of Magellan, but there are credible reports of its presence on the mainland (Wallem et al., In Press). This invasion wave has traveled at 3-6 km/year depending on direction, over the last 60 years. The population growth of the American beaver has been so explosive that both the Argentinean and Chilean governments are attempting to control this species, primarily because the beaver causes extensive damage to the native *Nothofagus* forests (Iriarte et al. 2005).

### European rabbit, *Oryctolagus cuniculus*

European rabbits were first introduced to central Chile in 1884, to an island in a lake (Cauquenes), from where they escaped when a drought enabled connection of the island with the mainland (Jaksic 1998). From this release point, rabbits expanded their distribution toward the north and south of the country. They also crossed the Andean Ranges toward Argentina and have continued their expansion there (Jaksic et al. 2002). This invasion wave has traveled at 7-16 km/year over the last 120 years. The negative effects of European rabbits on native vegetation and flora, both herbaceous and shrubby, are well documented (Jaksic 1998). The rabbit forestalls secondary succession of the

Chilean scrublands, rendering them more open and favoring their invasion by invasive herbs, mostly of Eurasian origin. At the same time, the case of the European rabbit is one of the most convincing examples of a positive effect of some invaders (Jaksic 1998). It has been documented that in less than 16 years, the native Culpeo fox (*Pseudalopex culpaeus*), Chilean eagle (*Geranoaetus melanoleucus*), and Magellan horned owl (*Bubo magellanicus*) have substantially increased their consumption of European rabbits. Therefore, the addition of this new prey resource can be considered beneficial for at least three native endangered predators.

## PROSPECTS FOR COLLABORATION

I foresee three main topics on which collaborative work could be initiated between US, Canadian, and Chilean researchers. First, invasion pathways, spread rates, and environmental impact of invaders could be studied. Parallel studies could be conducted on African clawed frog (Chile versus California) and European rabbit (Chile versus Hawaii). Second, the comparative ecology of invaders could be studied. For instance, African clawed frog (Chile versus California), California quail (Chile versus California), American beaver (Chile versus northeastern US and southeastern Canada) and European rabbit (Chile versus Hawaii) could be compared. Third, genetic diversity of populations of invaders could be studied. For instance, African clawed frog (Chile versus California versus South Africa), California quail (Chile versus California), American beaver (Chile versus northeastern US and southeastern Canada) and European rabbit (Chile versus Hawaii) could be compared. Chile's Center for Advanced Studies in Ecology and Biodiversity could be a focal point in coordinating such research efforts.

## LITERATURE CITED


FIGUEROA, J. A., S. A. CASTRO, P. A. MARQUET, AND F. M. JAKSIC. 2004. Exotic plant invasions to the mediterranean region of Chile: causes, history and impacts. Revista Chilena de Historia Natural 77:465-483.

IRIARTE, J. A., G. A. LOBOS, AND F. M. JAKSIC. 2005. Invasive vertebrate species in Chile and their control and monitoring by governmental agencies. Revista Chilena de Historia Natural 78:143-154.

JAKSIC, F. M. 1998. Vertebrate invaders and their ecological impacts in Chile. Biodiversity and Conservation 7:1427-1445.

JAKSIC, F. M., J. A. IRIARTE, J. E. JIMÉNEZ, AND D. R. MARTÍNEZ. 2002. Invaders without frontiers: cross-border invasions of exotic mammals. Biological Invasions 4:157-173.

LOBOS, G. A. AND F. M. JAKSIC. 2005. The ongoing invasion of African clawed frogs (*Xenopus laevis*) in Chile: causes of concern. Biodiversity and Conservation 14:429-439.

SWARTH, H. S. 1927. Valley Quail imported from Chile. Condor 29: 164.

WALLEM, P. K., C. G. JONES, P. A. MARQUET, AND F. M. JAKSIC. In Press. Identificación de los mecanismos subyacentes a la invasión de *Castor canadensis* (Rodentia) en el archipiélago de Tierra del Fuego, Chile. Revista Chilena de Historia Natural.

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